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Monitoring and Assessment Techniques

THE ST. ALBANS BAY WATERSHED RCWP: A CASE STUDY OF MONITORING AND ASSESSMENT

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ABSTRACT

Excessive nutrients from a municipal point source and agricultural nonpoint sources have impaired the use of St. Albans Bay of Lake Champlain in Vermont. A comprehensive monitoring and evaluation approach is evaluating the effects of agricultural Best Management Practices (BMP's) on the quality of bay and tributary waters as part of the Rural Clean Water Program (RCWP). Monitoring techniques include edge-of-field paired watersheds, in-stream trend stations, bay trend sampling, and land use tracking. Related short-term studies are investigating bay circulation patterns, bay sediment phosphorus content and release, biological indicators, and the role of a wetland in treating both point and nonpoint source nutrients. Each monitoring technique and its associated assessment methods are described through project results. The comprehensive monitoring approach is designed to identify overall programmatic effects in complex watersheds.

INTRODUCTION

The St. Albans Bay Watershed is one of the experimental Rural Clean Water Programs (RCWP) projects designed to improve water quality through agricultural best management practices (BMP). St. Albans Bay has been degraded by excessive algal and macrophyte growths and elevated coliform counts (Vt. Agency Environ. Conserv., 1977). Abundant nutrients in the bay, which are causing the accelerated eutrophication, come from both point and nonpoint sources. Recently, Johnson (1985) estimated that at least 37 percent of the phosphorus and 48 percent of the nitrogen originated from nonpoint sources. Improper animal waste management and cropping practices have been identified by the Soil Conservation Service (1981) as being primarily responsible for excessive nonpoint nutrient loading to the bay.

In 1981, implementation of agricultural BMP's began with Federal cost-sharing through RCWP to control nonpoint sources of nutrients and sediment. Concurrent with the agricultural nonpoint source control strategy is a comprehensive water quality monitoring and evaluation project to determine the effects of BMP's on water quality.

Numerous techniques have been used to assess the effect of land treatment on water quality. Listed in order of increasing distance from the source, these techniques include: runoff plots; fields; single, paired, and multiple watersheds; and larger, mixed land use watersheds (Striffler, 1965; Hewlett et al. 1969; Ponce, 1980; Clausen and Brooks, 1983). Advantages and disadvantages of these techniques have been described (Striffler, 1965; Hewlett et al. 1969; Clausen and Brooks, 1983). One of the greatest challenges facing water quality data analysts is the interpretation of water quality changes in streams receiving nutrients from large complex watersheds.

This paper describes the monitoring and assessment techniques being used in the St. Albans Bay RCWP and discusses current findings.

STUDY AREA

The 13,500 ha St. Albans Bay watershed is located in northwestern Vermont, 40 km north of Burlington (Fig. 1). Agriculture is the dominant land use in the watershed (68 percent); corn and hay are the principal crops. Forests cover 22 percent of the area, and urban areas and roads account for the remaining 10 percent. There are 100 dairy farms in the watershed averaging 134 ha and 95 animal units.

Watershed soils include loam (51 percent), half of which is poorly drained, silt and clay (27 percent), rock outcrop (15 percent), and sand (7 percent) (Soil Conserv. Serv. 1979). These soils formed on glacial till or lacustrine deposits.

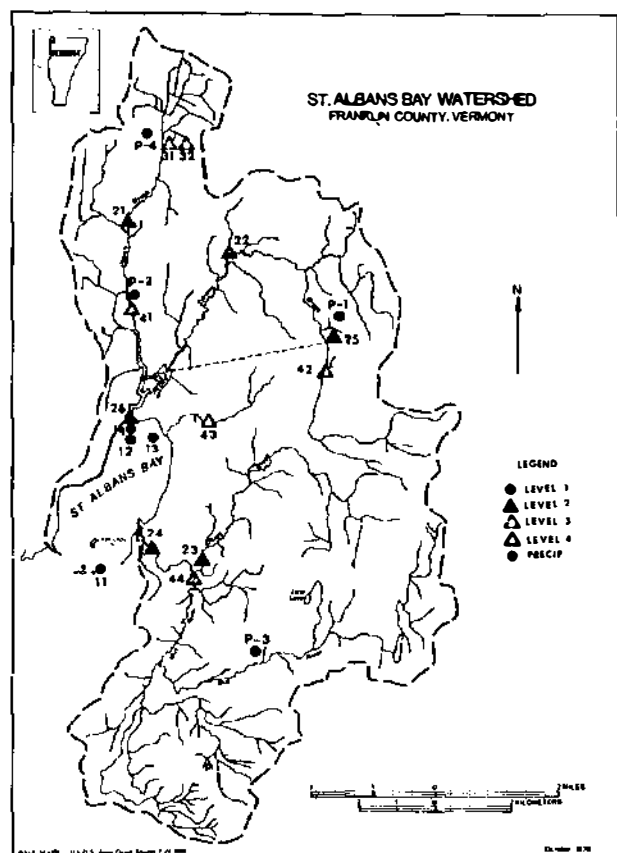


Figure 1.—Map of the St. Albans Bay watershed showing sampling locations.

The mean annual precipitation is 845 mm, and occurs mainly in the summer. The climate is considered to be the cool, continental type with a mean annual temperature of 7.3°C. Average annual snowfall is 1,560 mm (Soil Conserv. Serv. 1979).

Four major tributaries drain the watershed into St. Albans Bay: Jewett Brook, Stevens Brook, Rugg Brook, and Mill River (Fig. 1). The city of St. Albans' secondary wastewater treatment plant discharges to Stevens Brook wetland at the head of the bay.

METHODS

Sampling Design

To document water quality changes, several levels of water quality sampling have been conducted since 1981. Level 1 involves bay sampling at four stations, 20 times each year (Fig. 1). At each station, samples are collected at the 0.5 m depth and 1.0 m from the bottom. Level 2 includes instream sampling at the four tributaries and the St. Albans' wastewater treatment plant. At each of the five Level 2 stations, samples are automatically collected at 8-hour intervals using ISCO refrigerated samplers and combined into two 48-hour and one 72-hour composites each week. During stormflow periods, each 8-hour sample is analyzed discretely. Flow is measured continuously at each station using ISCO bubbler-type stage recorders. Three standard, weighing-bucket gauges are used to measure watershed precipitation.

Level 3 involves edge-of-field sampling to evaluate changes in the quality of runoff associated with best manure management. A paired watershed design was used where two field watersheds received best manure management during a 2-year calibration period, and then one

field received winter-spread manure during the treatment period. The control field was 0.9 ha and the treatment field was 1.9 ha. The treatment field received 8,925 kg/ha of liquid manure spread during winter 1984. Calibration and treatment regressions were based on paired daily concentration, discharge, and mass export values.

Level 4 sampling is conducted at four other stream locations in the watershed (Fig. 1) to characterize additional tributaries and to isolate subwatersheds. Grab samples are collected an average of once every 20 days on randomly selected dates.

Sample Analysis

All samples are analyzed for turbidity; total and volatile suspended solids; total and orthophosphorus; and total Kjeldahl, ammonia, and nitrate + nitrite nitrogen, according to standard methods (U.S. Environ. Prot. Agency, 1983). In situ measurements are made at all bay stations of temperature, dissolved oxygen, and Secchi disk. Weekly grab samples are analyzed for pH, conductance, fecal coliform, and fecal streptococcus. St. Albans Bay samples are also analyzed for chlorophyll *a*.

Related Studies

In addition to the long-term monitoring there have been separate investigations of stream biological characteristics (LaBar, 1984), bay circulation (Laible, 1983), and bay and wetland sediments (Drake, 1984). An extensive land use monitoring effort is described in detail in a companion paper (Hopkins and Clausen, 1985).

RESULTS AND DISCUSSION

BMP Implementation Status

The goals of the RCWP were to manage 75 percent of the 6,174 critical hectares in the watershed (lands receiving animal waste or fertilizer), and to treat a number of critical sources by using animal waste and fertilizer management, conservation cropping systems, and stream protection. Currently, approximately 90 percent of this goal has been achieved (Table 1). The major BMP is to provide for animal waste storage during the winter months to prevent daily manure spreading on snow covered or frozen soils. Under the Agricultural Conservation Program (ACP), two manure storage structures have been built and additional areas in conservation cropping supplement the RCWP.

Table 1.—BMP Implementation status for the St. Albans Bay watershed RCWP.

Year	Contracted Farms (No.)	Critical Areas Contracted (ha)	Manure Storage (No.)	Conservation Cropping area (ha)
1981	21	1,577	7	357
1982	18	1,314	21	1,200
1983	11	908	9	161
1984	6	398	5	550
Total	56	4,197	42	2,268
Goal	84	4,831	64	2,590

¹This is a sample for a typical footnote in 6 point helvetica by 19 picas.

St. Albans Bay

A horizontal gradient in concentration is evident in St. Albans Bay. The north end of the bay has much higher concentration of sediment and nutrients as compared to the south end which opens into Lake Champlain (Fig. 2). This gradient is related to mixing between the main lake

and the bay (Laible, 1983). Chlorophyll *a* concentrations follow these nutrient gradients. The inner bay averages 31 $\mu\text{g/l}$ chlorophyll *a* and the outer bay averages 9 $\mu\text{g/l}$. The total phosphorus to total nitrogen ratio in the bay ranges from 6:1 to 33:1, indicating that the limiting nutrient may at times be either nitrogen ($\text{TN:TP} < 10$) or phosphorus ($\text{TN:TP} > 17$) (Smith, 1982).

Detection of trends in the bay will have to consider these gradients, and both the chemical and biological characteristics of bay waters. Time trends may be confounded with hydrological variability. However, the outer bay station may serve as a control for comparison with the inner bay station. Trends could then be identified as differences between regressions, using the inner bay data as the dependent treatment variable.

Tributary Streams

Mean concentrations of solids, phosphorus, and nitrogen for the Level 2 tributary stations show both annual variability and differences among watersheds (Fig. 2). Annual precipitation for the 1982–83 water year was near normal (859 mm) while precipitation during 1983–84 was 30 percent above normal (1,094 mm). Although trends over 2 years of sampling mean little in water quality interpretations, observed concentrations do identify critical watersheds. For example, Jewett Brook, which has 87 percent agricultural land use, has elevated concentrations of phosphorus and nitrogen compared to other watersheds (Fig. 2). Mass exports in Jewett Brook are also quite high; during 1983–84 total phosphorus export was 6.7 kg ha⁻¹ yr⁻¹, over 20 times the average export from agricultural watersheds in the eastern United States (Omernik, 1976). The Jewett Brook Watershed has the most BMP's and therefore the potential for showing the greatest water quality changes during the project.

Edge-of-Field

The effects of winter-spread manure on field runoff concentrations are summarized in Figure 3. The dark bars represent the differences between the concentrations predicted by the calibration equation and the mean concentration observed during treatment. Winter spreading increased the concentrations of total P, ortho-P, total Kjeldahl-N, and ammonia-N, but total suspended solids decreased significantly ($p = 0.001$). After spreading manure in the winter, increased concentrations of phosphorus and nitrogen have been reported previously based on plot studies (Hensler et al. 1970; Minshall et al. 1970; Klausner et al. 1976). The reduction in suspended solids concentrations has also been reported (Young and Holt, 1977; Young and Mutchler, 1976), presumably resulting from a mulching effect of animal wastes.

Winter manure application decreased surface runoff from the field (Fig. 4). Runoff may decline because applied manure increases soil infiltration (Khaleel et al. 1981; Zwerman et al. 1970). The decrease in runoff together with reductions in suspended solids resulted in a decreased mass export of total suspended solids by one-half (Fig. 4).

Even though runoff decreased, phosphorus and nitrogen increased in runoff after winter manure applications (Fig. 4). Total phosphorus export increased 11 percent ($p = 0.08$), but orthophosphorus export increased by a factor of 15 ($p = 0.03$).

Based on the amount of manure applied to the field in the winter, 15 percent of the phosphorus and 17 percent of the nitrogen was lost in surface runoff. These losses are somewhat greater than the 95 percent retention of phosphorus and nitrogen of winter-applied manure reported by Klausner et al. (1976).

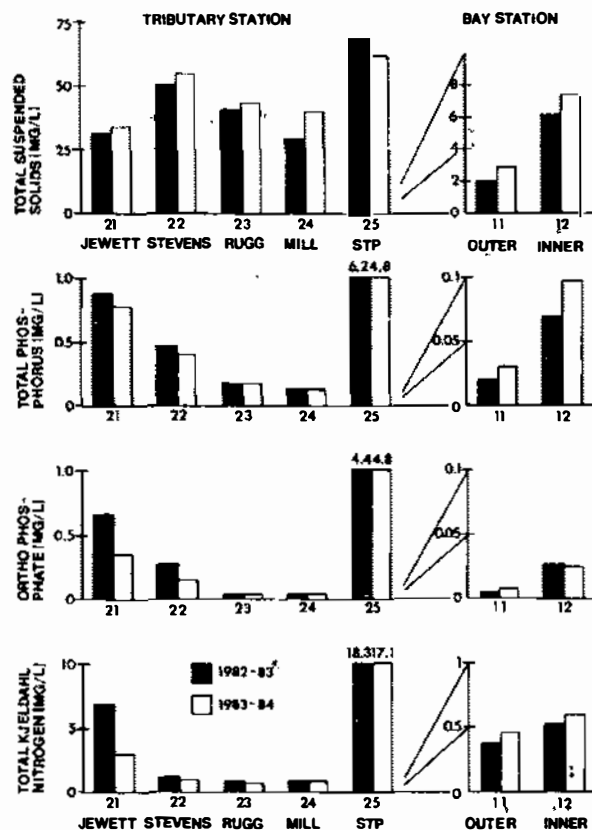


Figure 2.—Mean Concentrations of solids, phosphorus, and nitrogen at the tributary and bay trend stations for 2 years.

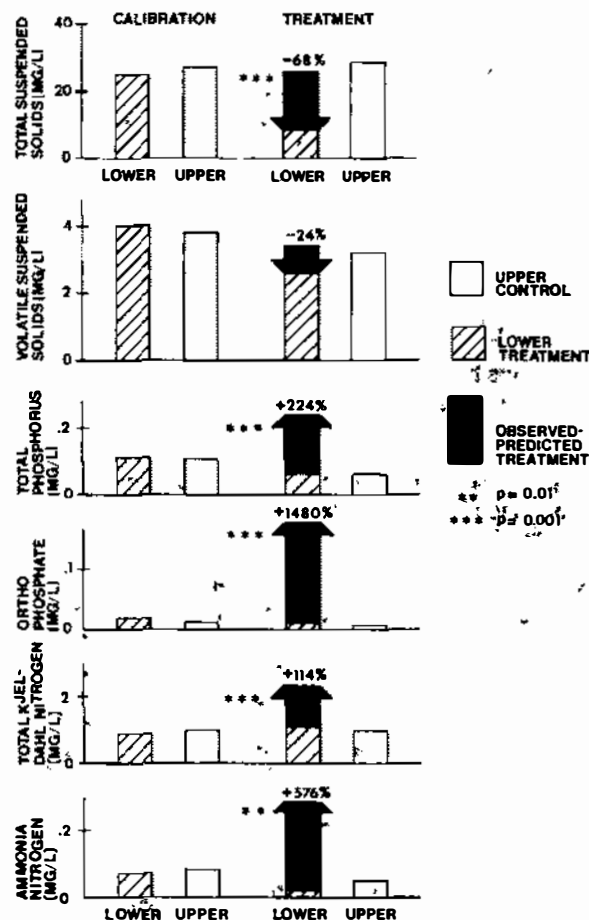


Figure 3.—Mean concentrations in runoff from the Larose farm paired watersheds.

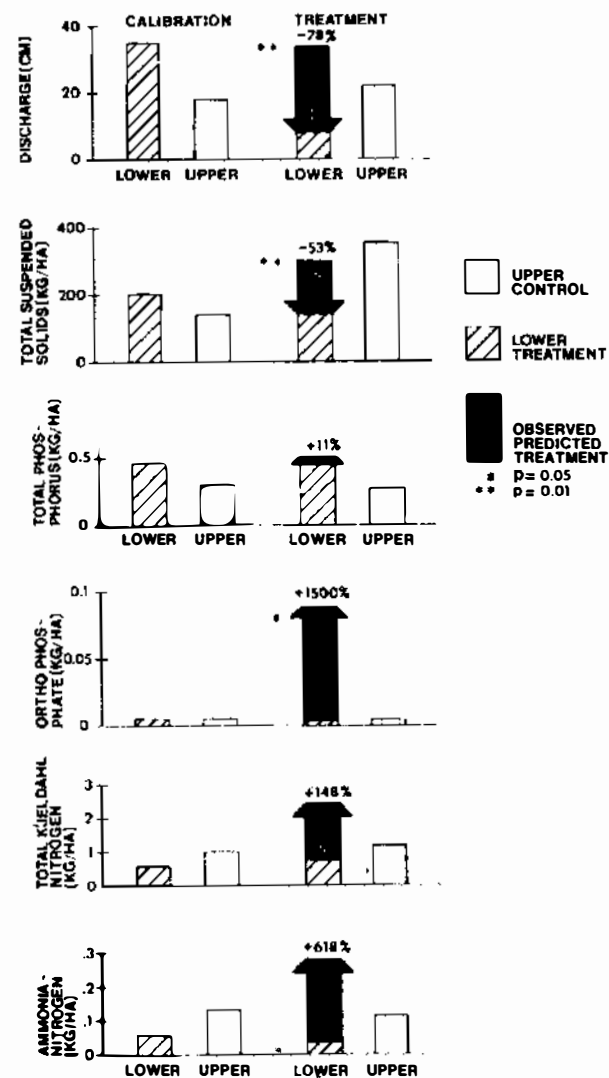


Figure 4.—Runoff and mass export from the Larose farm paired watersheds.

CONCLUSIONS

There has been insufficient time to evaluate water quality trends in the bay or its tributaries. However, the edge-of-field paired watershed experiment has shown, in a relatively short time, that proper animal waste management can reduce phosphorus and nitrogen concentrations and exports to receiving bodies of water.

Monitoring of water quality and agricultural activities will continue. Several techniques are available for water quality trend detection for long-term studies: (1) Time plots, (2) least squares regression, (3) comparisons of annual means, (4) Q-Q plots, (5) probability distribution functions, (6) paired watershed regression, (7) spectral analysis, and (8) time series analysis. Good descriptions of these methods appear in UNESCO (1978), Hirsch et al. (1982), and Montgomery and Reckhow (1984). As additional data become available, these trend assessment techniques will be applied to determine the changes in water quality associated with BMP implementation.

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ral Resources Conservation District, Vermont Extension Service, and the Vermont Agency of Environmental Conservation.

REFERENCES

- Clausen, J.C., and K.N. Brooks. 1983. Quality of runoff from Minnesota peatlands: II. A method for assessing mining impacts. *Water Res. Bull.* 19(5): 769-72.
- Drake, J.C. 1984. Sedimentological studies in St. Albans Bay and contiguous wetlands. *Annu. Rep. Dep. Geology, Univ. Vermont, Burlington.*
- Hensler, R.F., et al. 1970. Effect of method of manure handling on crop yields, nutrient recovery and runoff losses. *Trans. Am. Soc. Agric. Eng.* 13(3): 726-31.
- Hewlett, J.D., H.W. Lull, and K.G. Reinhart. 1969. In defense of experimental watersheds. *Water Res. Res.* 5(1): 306-15.
- Hirsch, R.M., J.R. Slack, and R.A. Smith. 1982. Techniques of trend analysis for monthly water quality data. *Water Res. Res.* 18(1): 107-21.
- Hopkins, R.B., and J.C. Clausen. This vol. Land use monitoring and assessment for nonpoint source pollution control. In *Perspectives on Nonpoint Source Pollution. Proc. Natl. Conf. N. Am. Lake Manage., Kansas City, MO. May 19-22, 1985.*
- Johnson, G.D. 1985. The effects of Stevens Brook Wetland on nutrient loading to St. Albans Bay. M.S. Thesis. Univ. Vermont, Burlington.
- Khaleel, R., K.R. Reddy, and M.R. Overcash. 1981. Changes in soil physical properties due to organic waste applications: a review. *J. Environ. Qual.* 10(2): 133-41.
- Klausner, S.D., P.J. Zwerman, and D.F. Ellis. 1976. Nitrogen and phosphorus losses from winter disposal of dairy manure. *J. Environ. Qual.* 5(1): 47-50.
- LaBar, G.W. 1984. St. Albans RCWP fish and benthos monitoring study. Final Report. School Nat. Resour., Univ. Vermont, Burlington.
- Laible, J.P. 1983. St. Albans Bay current and pollution transport study. Progress rept. Univ. Vermont, Burlington.
- Minshall, N.E., S.A. Witzel, and M.S. Nichols. 1970. Stream enrichment from farm operations. *Am. Soc. Civil Eng. J. San. Eng. Div.* 96(SA2): 513-24.
- Montgomery, R.H., and K.H. Reckhow. 1984. Techniques for detecting trends in lake water quality. *Water Res. Bull.* 20(1): 43-52.
- Oniarnik, J.M. 1976. The influence of land use on stream nutrient levels. EPA-600/3-76-014. U.S. Environ. Prot. Agency, Corvallis, OR.
- Ponce, S.L. 1980. Water quality monitoring programs. WSDG Tech. Pap. No. 00002. Watershed Systems Development Group, Forest Serv., U.S. Dept. Agric., Fort Collins, CO.
- Smith, V.H. 1982. The nitrogen and phosphorus dependence of algal biomass in lakes: an empirical and theoretical analysis. *Limnol. Oceanogr.* 27(6): 1101-12.
- Soil Conservation Service. 1979. Soil Survey of Franklin County, Vermont, U.S. Dept. Agric. Burlington, VT.
- _____. 1981. Rural Clean Water Program St. Albans Bay Project. Plan of Work. U.S. Dept. of Agric. Franklin County, VT.
- Striffler, W.D., 1965. The selection of experimental watersheds and methods in disturbed forest areas. Pages 464-473 in *Symp. Budapest. Internat. Assn. Scientific Hydrol. Publ.* No. 66.
- UNESCO. 1978. Water quality surveys—a guide for the collection and interpretation of water quality data. IHD-WHO Work. Group Qual. Water. UNESCO/World Health Organ., Geneva, Switzerland.
- U.S. Environmental Protection Agency. 1983. Methods for Chemical Analysis of Water and Wastes. EPA-600/4-79-020. Off. Res. Dev., Cincinnati, OH.
- Vermont Agency of Environmental Conservation. 1977. Nutrient loading to Shelburne Bay and St. Albans Bay, Lake Champlain, Vermont 1975-1976. Montpelier.
- Young, R.A., and R.F. Holt. 1977. Winter-applied manure: effects on annual runoff, erosion, and nutrient movement. *J. Soil Water Conser.* 32(5): 219-222.
- Young, R.A., and C.K. Mutchler. 1976. Pollution potential of manure spread on frozen ground. *J. Environ. Qual.* 5(2): 174-9.
- Zwerman, P.J., et al. 1970. Rates of water infiltration resulting from applications of dairy manure. Pages 263-270 in *Relationships of Agriculture to Soil and Water Pollution. Proc. 1970 Cornell Univ. Conf. on Agric. Waste Manage.*

LAND USE MONITORING AND ASSESSMENT FOR NONPOINT SOURCE POLLUTION CONTROL

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ABSTRACT

Varying soil characteristics, land use patterns, the relative timing of agricultural practices, and hydrologic events complicate quantifying relationships between agricultural land use and surface water quality. In two Vermont watersheds where the effects of best management practice (BMP) implementation on water quality are continuously monitored, land use and agricultural activities are being monitored on a field-by-field level. The land use data are entered in a computerized Geographic Information System (GIS), and the results mapped. Correlation and stepwise regression techniques related weekly land use activities for one subwatershed to surface water quality. Comparisons of water quality to agricultural land use were based on proximity to surface drainage and whether activities had occurred on runoff-producing zones. Manure application on Soil Hydrologic Group D was significantly related to stream total phosphorus concentration ($r = .62$) when manure was accumulated between runoff events. A predictive equation developed explained 55 percent of the variation in total phosphorus concentration. GIS offers the potential to inventory critical sources of nonpoint source pollution and identify changes in water quality from agricultural land use and BMP's.

STUDY AREA

The 1,333 ha Jewett Brook watershed in northwestern Franklin County, Vermont, was selected for the study (Fig. 1). Land use is predominantly agricultural (87 percent) with the remainder woodland or residential. The 16 dairy farms in the watershed average 65 ha, with herds ranging from 32 to 260 animals. The average herd size is approximately 125 animals.

Within the Champlain Lowland physiographic unit, the Jewett Brook watershed has irregular topography with rolling hills. Over two-thirds of this area has soil formed on lacustrine deposits; other soils formed on glacial till. Approximately 50 percent of the watershed has poorly drained silt and clay soils; another 42 percent has poorly drained loam or sand; and only 8 percent of the area has well-drained loam or sand (Soil Conserv. Serv. 1979).

The climate of the watershed is influenced by the presence of Lake Champlain to the west and south and by its northern latitude ($44^{\circ}47'26''$). Long-term average winter temperature is -7°C , and average summer temperature is 20°C . The average last spring freeze is expected May 2 and the first autumn freeze by Oct. 13. Approximately 61 percent of the total yearly precipitation falls in April through September, with August the wettest month.

INTRODUCTION

The relationship between land use and water quality has been the subject of much research in the last 10 years. It is generally accepted that as the percent of agricultural land in a watershed increases, concentrations of sediment and nutrients in streams draining these areas also increase (U.S. Environ. Prot. Agency, 1974; Dillon and Kirchner, 1975; Smolen et al. 1975; Omernik, 1976, 1977; Hill, 1981). The proximity of agricultural lands to streams within a watershed may also influence nutrient contributions in runoff (Kunkle, 1970; Uttormark et al. 1974). Dunne (1969), and Lake and Morrison (1977) report that large nutrient losses in runoff may originate from areas of low infiltration potential or high soil saturation. These areas have been termed runoff-producing zones.

Greatest stream nutrient concentrations have been linked to spring stormflow periods when cultivation is active and vegetative cover is poor (Dornbush et al. 1974; Dendy, 1981; McDowell et al. 1981), but this relationship may be caused solely by increased discharge in the spring, rather than agriculturally induced.

Agricultural activities (e.g. nutrient applications, cultivation) should influence stream water quality, with activities in runoff-producing zones and near streams having a greater effect than those elsewhere. These relationships have not been temporally or spatially examined. The primary purpose of this study is to relate the location and timing of agricultural activities to stream water quality.

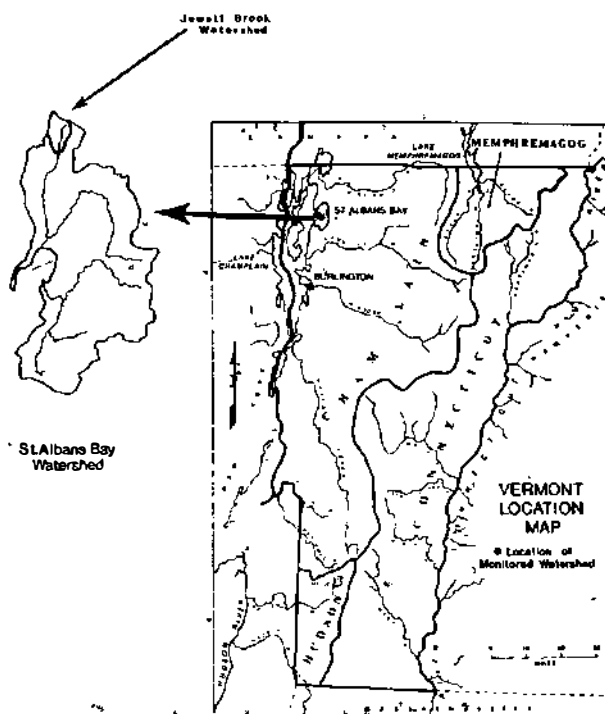


Figure 1.—Vicinity map for project location.

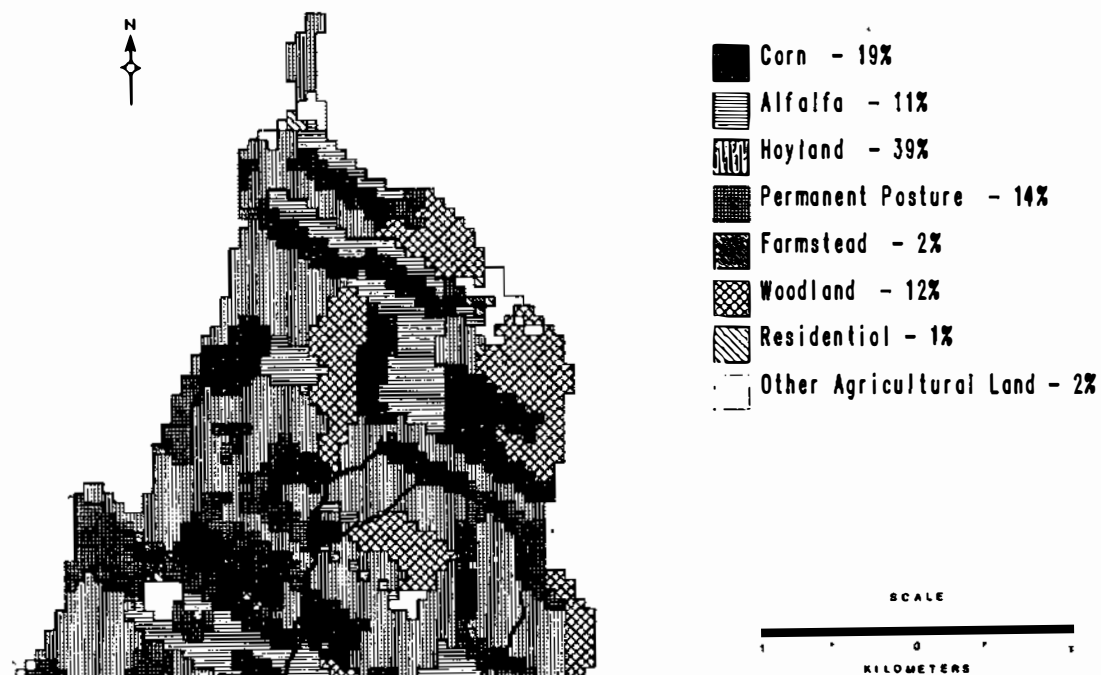


Figure 2.—Land use for Jewett Brook watershed.

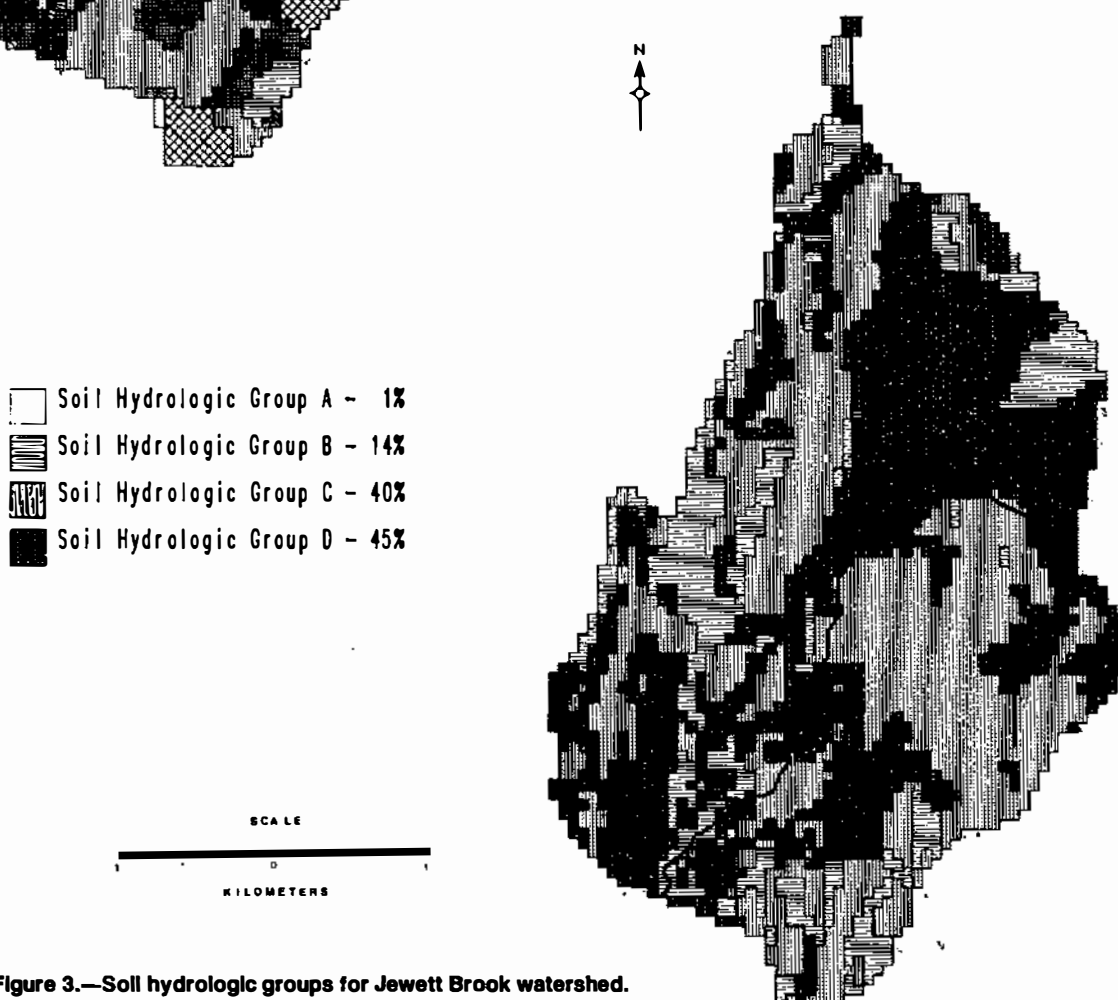


Figure 3.—Soil hydrologic groups for Jewett Brook watershed.

(10.0 cm). On the average, December receives the greatest amounts of snowfall (49.2 cm) (Soil Conserv. Serv. 1979).

METHODS

Maps of the watershed were prepared at a scale of 1:10000. Land use and farm and field boundaries were identified during interviews with each landowner. Soil types and characteristics were obtained from the Franklin County Soil Survey (Soil Conserv. Serv. 1979). Streams and drainage ditch locations were identified using topographic maps and aerial photographs. Elevation and watershed boundaries were obtained from USGS 7.5' topographic maps (U.S. Geolog. Surv. 1972). Data were entered into a computerized Geographic Information System (GIS) using a 0.404 ha cell grid overlay. Figures 2 and 3, generated by the GIS, show watershed land use and soil hydrologic classifications, respectively.

Land Use Monitoring

Land use and areas receiving agricultural activities were recorded onto field logs that had been distributed to each landowner within the watershed. Agricultural activity data were recorded from January to December, 1983. Data included the date, amount, location, and method of commercial fertilizer and animal waste application, areas that had been plowed and cultivated, and fields where crops had been harvested. Information was gathered during January, June, and December. Data were mapped using the GIS.

Computerized geographic overlays were performed using the GIS. Overlays were created with weekly land use data, soil hydrologic classifications, and the area within 63 meters of the brook and drainage network. Runoff-producing zones were areas associated with Soil Hydrologic Group D (those soils having high runoff potential and low infiltration rates).

Table 1.—Correlations (r) between mean weekly runoff concentrations (mg/l) and weekly hydrologic variables.

Water Quality Parameters	Total precipitation (cm)	Mean discharge (m^3/sec)
Total phosphorus	.07	.37 ²
Orthophosphorus	.06	.48 ²
Total Kjeldahl nitrogen	.06	.44 ²
Ammonia nitrogen	.02	.33 ¹
Total suspended solids	.32 ¹	.61 ²
Volatile suspended solids	.24	.68 ²

¹Indicates significance at $P = 0.05$

²Indicates significance at $P = 0.01$

Table 2.—Correlations (r) between mean weekly runoff concentrations (mg/l) and weekly manure applications (mT).

	Applied to:			
	Total watershed	Hydrologic group D	Within 63 m	Within 63 m on D
Total phosphorus	.47 ²	.38 ²	.39 ²	.20
Orthophosphorus	.26	.15	.16	.10
Total Kjeldahl nitrogen	-.04	-.11	-.15	-.12
Ammonia nitrogen	-.18	-.23	-.26	-.32 ¹
Total suspended solids	.18	.23	.18	.15
Volatile suspended solids	.14	.20	.14	.21

¹Indicates significance at $P = 0.05$

²Indicates significance at $P = 0.01$

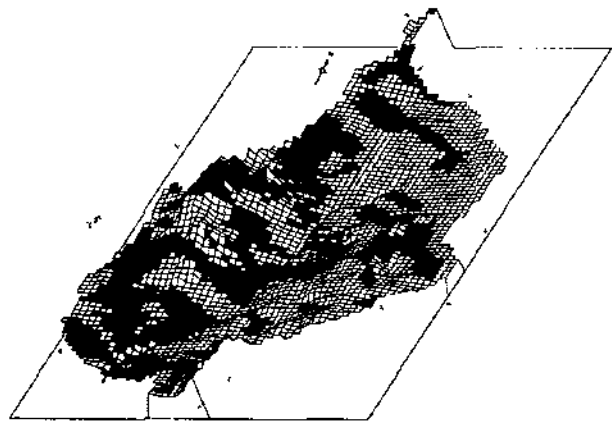


Figure 4.—Manure applications (in black) in Jewett Brook watershed during 1983.

Water Analysis

Streamflow quantity and quality were continuously monitored at the watershed outlet. Two 48-hour and one 72-hour composite water samples were collected each week for 52 weeks. Samples were analyzed for total suspended solids, volatile suspended solids, total Kjeldahl nitrogen, ammonia nitrogen, total phosphorus, and orthophosphorus according to Standard Methods (1980; U.S. Environ. Prot. Agency, 1983). A detailed description of the comprehensive water quality monitoring program can be found in Cassell et al. (1983).

RESULTS AND DISCUSSION

Weekly Activities

Weekly mean stream concentrations were positively correlated with weekly mean discharge but were not generally related to total weekly precipitation (Table 1). Suspended solids concentrations were strongly related to discharge. This positive relationship between discharge and streamflow concentrations is characteristic of diffuse sources of nutrients and sediment (Novotny and Chesters, 1981).

Weekly mean total phosphorus concentrations in streamflow were positively correlated with weekly manure applications within the watershed (Table 2). In Figure 4, the darkened areas represent manure applications in the watershed during 1983. Correlations generally decreased when considering smaller components of the watershed as compared to applications throughout the watershed. Considering manure applications on the greatest runoff-producing zones (soil hydrologic group D) did not improve correlations. Manure applications in closer proximity (63 m) to stream courses were not as well related to

streamflow total P as were overall watershed manure applications.

Generally, poor correlations between land use changes and water quality were observed. For example, the correlation between total phosphorus concentrations and the percent of corn land was only -0.19 . A possible explanation may be that only 3 percent of the watershed changed land use between corn, alfalfa, and hay during this 1-year study. Also, poor correlations were generally obtained between areas receiving field management and total phosphorus concentration (e.g., cultivation, $r = 0.26$). This lack of correlation resulted partially from the timing of activities. For example, 81 percent of the cultivation occurred during a 9-week period in the spring. During the remaining 43-week period, little or no cultivation occurred, whereas weekly stream concentrations fluctuated greatly.

Lagged Activities

Mean daily discharge rates were examined to estimate the weeks of stormflow. Weekly land use activity data were accumulated between stormflow periods and then compared to stream concentrations. This method of comparison assumes primary nutrient and sediment movement during stormflow.

When manure applications were accumulated between stormflow periods and compared to in-stream concentrations, stronger correlations resulted (Table 3). Total and orthophosphorus and total Kjeldahl nitrogen were positively correlated to applied manure using this lagging technique. Generally, manure applied throughout the watershed correlated better with stream concentrations than manure applied to runoff-producing zones. Proximity did not appear to greatly influence these relationships. The relationship between total phosphorus and accumulated manure applied is shown in Figure 5.

Since both stream discharge and manure applications were related to stream phosphorus concentrations, multiple regression was used in an attempt to explain more of the variation in stream concentrations. The best prediction of total phosphorus concentration ($P = 0.01$) resulted from using manure applied on Group D soils and total suspended solids concentrations in runoff (Log total P = $0.15 \text{ Log Manure on D} + 0.34 \text{ Log total S.S.} - 1.09$; multiple $r^2 = 0.55$). This relationship suggests that manure applications to low infiltration rate soils combined with suspended solids in runoff are the primary variables influencing stream phosphorus concentrations. On the average, 38 percent of the instream total phosphorus concentrations were in particulate form. During storm events, up to 90 percent of the total phosphorus was particulate. Surprisingly, discharge did not significantly add to the regression.

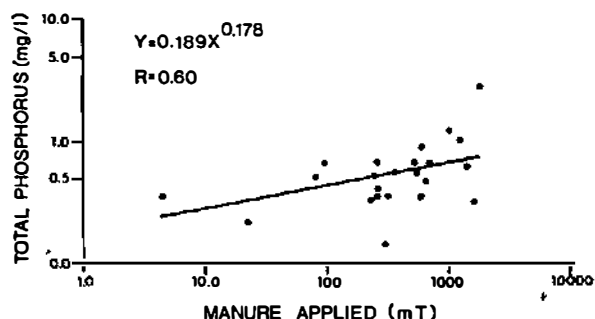


Figure 5.—Total phosphorus (mg/L) and manure applied (mT) between runoff events for the Jewett Brook watershed.

CONCLUSIONS

The concentrations of weekly total phosphorus in Jewett Brook were positively related to mean weekly stream discharge and the weekly amounts of manure applied to the watershed. Considering manure applications adjacent to the brook did not improve simple correlation relationships.

Accumulated manure applied between stormflow events improved correlations with stream phosphorus concentrations. However, the proximity of these applications did not greatly improve relationships.

Multiple regressions suggested that manure applications on low infiltration rate soils and suspended solids in runoff explained variation in stream phosphorus concentrations more than other land use and hydrologic variables.

To better link land-use activities to stream water quality, one might consider mass export rather than just mean concentrations using the lagging techniques described. Shorter time intervals than weekly might also improve relationships. Finally, quantify differences between seasons, land use should be monitored for more than 1 year.

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Table 3.—Correlations (r) between mean weekly runoff concentrations (mg/l) and accumulated manure between runoff events (mT).

	Applied to:			
	Total watershed	Hydrologic group D	Within 63 m	Within 63 m on D
Total phosphorus	.60 ²	.62 ²	.60 ²	.57 ²
Orthophosphorus	.44 ²	.43 ¹	.43 ¹	.38
Total Kjeldahl nitrogen	.52 ¹	.50 ¹	.51 ¹	.48 ¹
Ammonia nitrogen	.35	.34	.30	.25
Total suspended solids	.16	.24	.19	.19
Volatile suspended solids	.19	.28	.19	.28

¹Indicates significance at $P = 0.05$

²Indicates significance at $P = 0.01$

REFERENCES

- Cassell, E.A., et al. 1983. St. Albans Bay watershed comprehensive water quality monitoring and evaluation. Rep. No. 1, Background Infor. Vt. Water Resour. Res. Center, Univ. Vermont, Burlington.
- Dendy, F.E. 1981. Sediment yields from a Mississippi delta cotton field. *J. Environ. Qual.* 10(1): 482-6.
- Dillon, P.J., and W.B. Kirchner. 1975. The effects of geology and land use on the export of phosphorus from watersheds. *Water Res.* 9:135-48.
- Dornbush, J.N., J.R. Andersen, and L.L. Harms. 1974. Quantification of Pollutants in Agricultural Runoff. U.S. Environ. Prot. Agency Environ. Prot. Technol. Ser. EPA-660/2-74-005.
- Dunne, J. 1969. The significance of 'partial-area' contributions to storm runoff for the study of sources of agricultural pollutants. Unpubl. mss., Agric. Res. Serv. Danville, VT.
- Hill, A.R. 1981. Stream phosphorus exports from watersheds with contrasting land uses in southern Ontario. *Water Res. Bull.* 17(4): 627-34.
- Kunkle, S.H. 1970. Sources and transport of bacterial indicators in rural streams. Pages 105-132 in *Proc. Symp. Interdisciplinary Aspects of Watershed Management. Irrigation and Drainage Div. Am. Soc. Civil Eng.*
- Lake, J., and J. Morrison. 1977. Environmental impact of land use on water quality. EPA-905/9-77-007-B. Final rep. Black Creek Proj: U.S. Environ. Prot. Agency, Washington, D.C.
- McDowell, L.L., et al. 1981. Toxaphene and sediment fields from a Mississippi delta watershed. *J. Environ. Qual.* 10(1): 121-25.
- Novotony, V., and G. Chesters. 1981. *Handbook of Non-Point Pollution, Sources and Management.* Van Nostrand Reinhold Co. New York.
- Omernik, J.M. 1976. The influence of land use on stream nutrient levels. *Ecol. Res. Ser. EPA-600/3-76-014.* U.S. Environ. Prot. Agency, Washington, D.C.
- _____. 1977. Non-point source—stream nutrient level relationships: a nationwide study. *Ecol. Res. Ser. EPA-600/3-77-105.* U.S. Environ. Prot. Agency, Washington, D.C.
- Smolen, M.D., M. Rashake, and V.O. Shanholtz. 1975. Effect of agricultural drainage on water quality. *Am. Soc. Agric. Eng. Pap. No. 75-2561.*
- Soil Conservation Service. 1979. Franklin County soil survey. Nat. Coop. Soil Surv. U.S. Dept. Agric.
- Standard Methods for the Examination of Water and Wastewater. 1980. 15th ed. Am. Pub. Health Assn., Washington, D.C.
- U.S. Environmental Protection Agency. 1974. Relationships between drainage area characteristics and non-point source nutrients in streams. *Natl. Eutrophication Surv. Work. Pap. No. 25.*
- _____. 1983. Methods for chemical analysis of water and wastes. EPA-600/4-79-020. Off. Res. Develop. Cincinnati, OH.
- U.S. Geological Survey. 1972. St. Albans Bay, Vermont..SW/4 St. Albans 15' Quadrangle. Reston, VA.
- Uttormark, P.D., J.D. Chapin, and K.M. Green. 1974. Estimating nutrient loadings of lakes from non-point sources. EPA-660/3-77-020. U.S. Environ. Prot. Agency, Washington, D.C.

APPROPRIATE DESIGNS FOR DOCUMENTING WATER QUALITY IMPROVEMENTS FROM AGRICULTURAL NPS CONTROL PROGRAMS

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ABSTRACT

Appropriate experimental designs are a function of the question to be answered. In the case of agricultural NPS control programs, the question is usually: How does BMP implementation affect the magnitude of pollutant concentrations or loads? This paper discusses the assumptions, analysis techniques, and advantages and disadvantages of three basic experimental designs that can be used in practical terms. Monitoring above and below an implementation site is generally more useful for documenting the severity of an NPS than for documenting BMP effectiveness. Time trend designs may be helpful; however, water quality trends are a result of complex interactions between land treatment, hydrology, and meteorologic factors. Accounting for these variables will therefore greatly increase the probability of documenting water quality improvements associated with BMP's. Paired watershed designs have the greatest potential for documenting improvements from BMP implementation because of the ability to control for meteorologic and hydrologic variability.

INTRODUCTION

A vast amount of information exists about best management practices (BMP's) for control of agricultural nonpoint sources (NPS). Most of this information, however, is from research efforts that considered only field plots or small watersheds. The investment of public funds to control nonpoint source pollution from agriculture requires that there be some assurance that nonpoint source pollution control programs be effective in protecting water quality. Hence, monitoring programs have been incorporated into many of these programs to verify that their application to the real world is, indeed, effective.

To evaluate the effectiveness of large-scale programs, such as the Rural Clean Water Program, projects (12,000–40,000 ha), requires a great deal of money. Therefore, data analysis should be planned and executed carefully following a clearly specified experimental design. Lack of an experimental design often results in wasted data collection efforts, and inconclusive results.

In this paper, we present and discuss three alternative experimental designs that are applicable to most nonpoint source control projects. The methodologies are applicable to surface and ground water studies that deal with BMP effects on pollutant concentrations, loads, or the frequency of standard violations. Most of our examples are presented in terms of surface water concentration, but only for convenience. This treatment is not rigorous statistically, but we have attempted to present useful suggestions and lay out some of the advantages, disadvantages, and assumptions associated with each design.

MONITORING DESIGNS AND ANALYSES

Before and After (Time Trends or Time Series Analyses) Uncorrected for Meteorological Variables

Definition, Advantages, and Disadvantages: The before and after design is generally characterized by monitoring one or more sites in a watershed over time to determine whether a change in water quality conditions has occurred. Agricultural nonpoint source control programs generally involve water quality monitoring over a period of several years below the agricultural nonpoint source to assess the concentration or loading changes associated with BMP implementation.

This design is the easiest to conduct with limited funds and personnel. Little coordination between land treatment and water quality monitoring personnel is required. In nearly all cases the entire project area can be monitored. There are no physical limitations to applying this basic design to any watershed.

A disadvantage is that sensitivity is low unless meteorologically related variables are measured (stream flow, precipitation, lake levels, ground water levels). Thus, it is difficult to attribute water quality changes to land treatment measures. A long monitoring period is needed to assess whether significant changes in water quality have occurred. This is due to the extreme hydrological and meteorological variability in most systems.

Appropriate Hypothesis, Data Requirements, and Assumptions: For conceptual clarity, all the hypotheses will be stated in the alternative rather than the null form. When meteorologic variables are not measured, the appropriate hypothesis is:

H_a: Mean annual (or seasonal) pollutant concentrations will decrease over time as BMP's are implemented.

The data needed to test this hypothesis are important. The sampling regimes should be similar for pre- and post-BMP implementation periods. Samples should be collected at equally spaced intervals or other predetermined schedules. It is important that sampling not be taken more frequently than scheduled. This allows pre- and post-BMP data to be compared with a minimum chance of sampling bias.

One assumption associated with this hypothesis is that every sample can be classified as belonging to either the pre- or post-BMP implementation period. If statistical tests are performed that divide the data into only these two groups, it is assumed that the level of BMP implementation is similar in each of the post-BMP years. Since this is

often not the case, these tests may produce conservative estimates of effects.

Hypothesis Test, Conclusions, and Interpretations:
The hypothesis can be tested using the Students t-test:

$$t_{\text{sample}} = \frac{(\bar{C}_{\text{pre}} - \bar{C}_{\text{post}})}{\sqrt{\frac{1}{y_{\text{pre}}} + \frac{1}{y_{\text{post}}} \frac{s_p^2}{n}}}$$

where n = the number of samples taken in each year or in each session if stratified, assumed constant

$$s_p^2 = \text{Pooled variance} = \frac{\sum_{i=1}^Y s_i^2}{Y}$$

y = the total number of years or seasons of monitoring

y_{pre} = the number of years or seasons pre-BMP

y_{post} = the number of years or seasons post-BMP

\bar{C}_{pre} = the mean of the pre-BMP concentrations.

\bar{C}_{post} = the mean of the post-BMP concentrations.

This t-sample statistic is compared to a t-table with $(Y \cdot n - Y)$ degrees of freedom. It should be noted that it may be advantageous to delete the interim time period if it can not be classified as pre- or post-BMP for this particular analysis.

An analysis that takes into account the cumulative nature of land treatment is the regression of concentration versus BMP application level. A significant negative slope suggests an improvement of water quality associated with BMP's. This approach does not require deleting data from intermediate years.

A third analysis that can be useful is generation of a Quantile-Quantile (Q-Q) plot. This analysis requires several steps. First, one generates a cumulative distribution of concentration for each site. This involves ranking by magnitude the concentration data and grouping it into percentiles. The mean for each percentile is calculated for both the pre- and post-BMP periods. These pairs are then plotted and the slope is tested to determine if it is significantly less than 1. An example of this plot is given in Figure 1. In this example a slope of less than 1 suggests a downward concentration trend.

Because uncontrolled variables such as flow have such a pronounced effect, often a downward concentration trend will not be observed. Even if a decrease in concentration is seen, no cause and effect relationship with BMP implementation level can be made. In a physical sense, there are four possible scenarios that may occur.

1. Mean flows increase; concentrations increase.
2. Mean flows increase; concentrations decrease.
3. Mean flows decrease; concentrations decrease.
4. Mean flows decrease; concentrations increase.

Of these four scenarios, there is generally only one (2) that provides strong evidence that BMP applications improved water quality. Also, without flow measurements, it is not possible to determine which of these four situations has occurred. Hence, without flow measurements, it is inevitable that a long-term monitoring program will be required to average out the fluctuations caused by stream flows, and to determine true effects of land treatment.

Before and After Time Trends Corrected for Stream Flows

Definition, Advantages, and Disadvantages: This design involves monitoring both concentration and flows over time at one or more sites in a watershed. Based upon

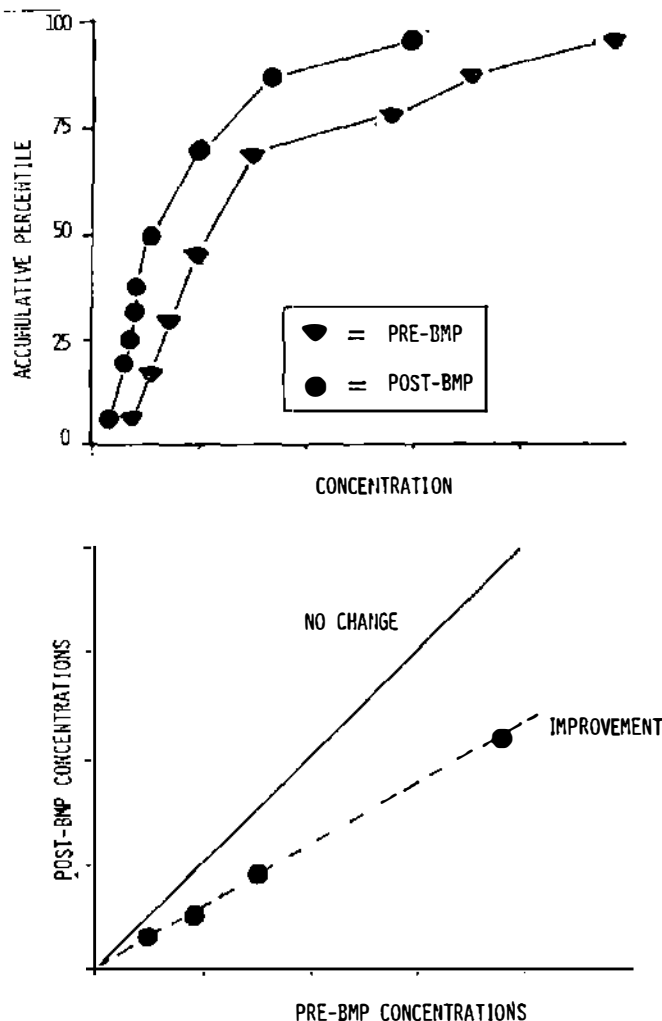


Figure 1.—An example of a Quantile-Quantile (Q-Q) plot derived from a plot of cumulative frequency distributions of concentration data from a pre- and post-BMP period.

previous studies, the variable with the greatest influence on surface water loads and concentrations is stream flow volume. (Froehlich, 1976; Johnson et al. 1974; McCool and Papendick, 1975). Thus, stream flows will be used in this and all subsequent examples that attempt to correct for meteorologic variations.

The basic advantages are the same as for the case just described. In addition, a stronger association with land treatment can be made. A long monitoring period is still needed to determine whether significant changes in water quality have occurred. Disadvantages are reduced, but unknown or unmeasured factors that occur during the project may still greatly reduce sensitivity.

Appropriate Hypothesis, Data Requirements and Assumptions: The hypothesis tested in this experimental design is:

H_a: Mean annual (or seasonal) pollutant concentrations will decrease over time when corrected for stream flows.

Flow-concentration pairs (concentration and flow measurements) need not be taken at equally spaced or predetermined time intervals. In fact, it can be seen from Figure 2 that the required data can be generated more efficiently if the monitoring is weighted toward periods of high flow. A wide range of flows is needed to establish a flow-concentration relationship, and the potential effects of BMP's are often greatest at high flows. Since the flow-concentration relationship often depends greatly upon whether the sample is taken during the rising or receding

limb of the hydrograph (Baker, 1985), it may be advisable to partition the data on this basis.

All the assumptions stated for the uncorrected, before and after design still hold. In addition, this design assumes that the BMP's will decrease pollutant concentrations more than they will reduce stream flows. In general, the assumption will hold for sediment and sediment-adsorbed pollutants, but may be in error for pollutants lost primarily in the dissolved phase of runoff. The pre- and post-BMP flow-concentration sample pairs need to reflect similar ranges in flows. If not, only the post-BMP data taken in the flow ranges present in the pre-BMP data should be used in the analyses.

Hypothesis Tests, Conclusions and Interpretations: Separate linear regressions of concentrations versus flows for the pre- and post-BMP periods can be performed. The slopes are compared for equality for the two periods as shown in Figure 2. From this analysis we can determine whether concentrations have changed over time for a given flow rate. With the establishment of a good flow-concentration relationship, the effects of BMP's can be distinguished under all four of the scenarios described. There may be a significant seasonal influence on the concentration-flow relationship. This source of variability in the data can be eliminated by partitioning the data by seasons. The cost of this partitioning, however, is a loss in the number of degrees of freedom (effective sample number), which decreases the sensitivity of the subsequent statistical tests.

Above and Below (Upstream-Downstream)

Definition, Advantages and Disadvantages: This experimental design involves sampling a flowing system over time above and below a potential nonpoint source. This has classically been the design used to monitor the effects of nonpoint source discharges to flowing systems.

The primary advantage of this approach is that it can account for upstream inputs to the area of interest. For agricultural nonpoint source projects, this will often be im-

portant for watersheds where the upper portions are in nonagricultural land uses. In addition, some irrigation management projects receive irrigation water that varies greatly in quality on an annual or seasonal basis. Perhaps the most common use of this design, however, is to document the location and magnitude of sources. As with the before and after design there is also the advantage that little or no coordination is required between the land treatment and water quality monitoring components of the project.

If the surface or ground water system originates within the nonpoint source area, there will be no suitable above sites. Also, the design provides only limited control for meteorologic variables, unless stream flow is monitored as described in the before and after design. In addition, it requires twice as many sampling sites as the before and after design to monitor an equivalent amount of the watershed area. The procedure may have low sensitivity because individual nonpoint source inputs are often small compared to background.

Appropriate Hypotheses, Data Requirements, and Assumptions: This design will generally provide information for testing two hypotheses: one concerning problem identification, and another concerning the effects of BMP's over time.

- Ha a. Agricultural pollutant concentrations will be higher downstream from a suspected agricultural nonpoint source as compared to upstream.
- Ha b. The difference between upstream and downstream pollutant concentrations will decrease over time as BMP's are applied.

Testing hypothesis a. requires paired concentration data above and below the potential nonpoint source over time during the pre-BMP period. For hypothesis b. the same paired data are needed for both the pre- and post-BMP periods.

The most important assumption for this design is that sampling is timed so that the same parcel of water is being sampled at the above and below sites. This requires some understanding of the hydrology system.

Hypotheses Tests, Conclusions, and Interpretations: For hypothesis a. to determine whether there is a significant concentration increase, a simple one-sided Student's t-test is used to determine whether the means of the paired differences between the upstream (C_{up}) and downstream (C_{down}) concentrations are different from zero.

$$t_{calc} = \frac{\bar{D}}{s_D}$$

where \bar{D} = the average of the paired differences,

$$\bar{D} = \frac{\sum_{i=1}^n (C_{up} - C_{down})}{n}$$

$$s_D = \frac{s_d}{\sqrt{n}}$$

In many cases, it is desirable to know what percentage of the pollutant concentration is attributable to the nonpoint source. The best estimate of this can be calculated from:

$$NPS \text{ Percentage} = \frac{n}{\sum} [(C_{down} - C_{up}) / C_{down}] * 100/n$$

To test hypothesis b., paired differences (D_i) must first be calculated for pre- and post-BMP periods ($D_i = C_{i down} - C_{i up}$). Then, each of the four analyses described for the before and after design can be used to test for water quality improvements associated with BMP implementation. Briefly, these include: (a) Student's t-test for determining

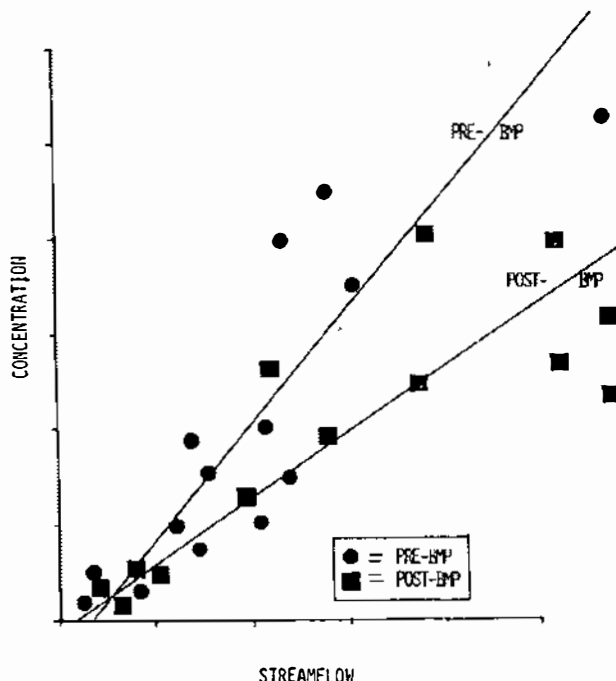


Figure 2.—An example of separate linear regression of concentration versus streamflows for a pre- and post-BMP period. Note that in this hypothetical example the data show a significant decrease in post-BMP concentrations when corrected for streamflow, even though the actual concentration mean is higher for the post-BMP period.

whether pre- and post-BMP mean concentrations are different, (b) Q-Q plots, (c) linear regression of D_i versus BMP implementation level, and (d) linear regressions of D_i versus flow for pre- and post-BMP periods to test for equality of flow-corrected D_i 's.

From testing hypothesis a, we can conclude whether the suspected agricultural nonpoint source is actually a significant contributor to an identified water resource impairment. From this, we can estimate the upper limit of how such improvement can be accomplished using BMP's.

For hypothesis b, the interpretations are very similar to those that can be made for the before and after design. In the cases where not all the water originates within the project area this experimental design allows trends to be established with more certainty than the before and after design, because of the corrections for incoming concentrations.

Paired Watersheds Design (Controlled-Experimental Design or Treated-Untreated Design)

Definition, Advantages, and Disadvantages: The design consists of monitoring downstream from two or more agricultural drainages where at least one drainage has BMP implementation, and at least one does not. This design ideally possesses the following characteristics: (a) simultaneous monitoring below each drainage, (b) monitoring at all sites prior to any land treatment (calibration period) to establish the relative responses of the drainages, and (c) subsequent monitoring, where at least one drainage area continues to serve as a control through the land treatment period, i.e., receives significantly less land treatment than the other drainage areas.

This design controls for meteorologic (and to some extent hydrologic) variability, minimizing the need for monitoring meteorological parameters. In most cases, water quality improvements related to BMP implementation can be documented within a much shorter time frame. In addition, this design provides stronger statistical evidence of the cause-effect relationship between agricultural nonpoint source control efforts and water quality changes.

A disadvantage of this design is that land treatment and water quality personnel must coordinate closely to match implementation efforts with monitoring and data analysis needs. For some projects it may be difficult to find adequately similar drainages. Close physical proximity is essential. Another disadvantage is the fact that control basins cannot receive as much land treatment, thus reducing the potential water quality improvement for the overall project area. This design is not intended to determine the location or severity of the nonpoint source.

Appropriate Hypothesis, Data Requirements, and Assumptions:

H_a: An agricultural drainage with BMP's applied will exhibit a decrease in pollutant concentrations over time, relative to an untreated agricultural drainage.

Site selection is crucial to this design. A similarity in hydrology and land use is desirable. Sampling from the watersheds should be conducted consistently (either simultaneously or separated by a constant time interval). Because concentration-flow relationships vary with rising or falling hydrograph limb, it is desirable to partition data on this basis.

It is assumed that paired watersheds have similar precipitation patterns, because of their geographic proximity. The hydrologic response of the paired watersheds should be consistent, even if actual concentrations are quite different because of differences in slope, soil type, cropping

	CONTROL BASINS	EXPERIMENTAL BASINS
CALIBRATION PERIOD	NO BMP's	NO BMP's
TREATMENT PERIOD	NO BMP's	BMP's APPLIED

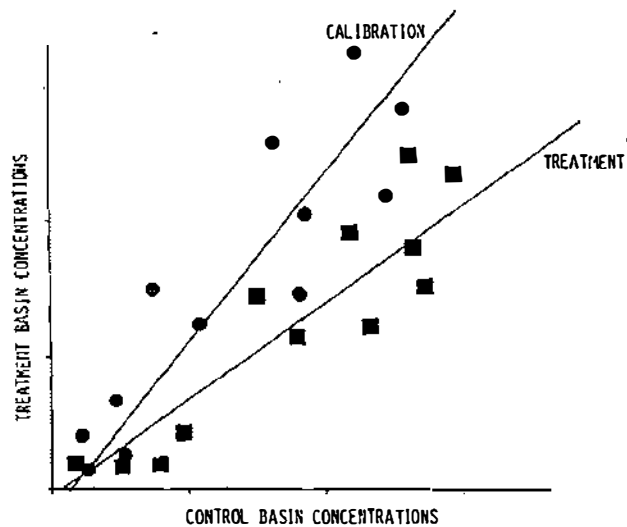


Figure 3.—An example of data analysis for the paired watersheds experimental design. If the predicted watershed value is significantly less during the treatment period as compared to the calibration period, a significant improvement in pollutant concentrations is indicated.

patterns, and other factors. It is assumed that BMP implementation levels can be measured accurately. Finally, the precipitation, stream flows, and cropping patterns should be at least somewhat similar for the calibration and treatment periods.

Hypothesis Tests, Conclusions, and Interpretations:

Linear regressions of the concentrations (or log concentrations) for the treatment versus the control watersheds for the calibration and land treatment periods can be performed (Fig. 3). A Student's t-test is performed to determine if the predicted treatment watershed values at the mean control watershed concentration decrease over time.

A decrease in the predicted treatment watershed values suggests a positive effect of BMP's on the water quality. This is stronger evidence of a cause-effect relationship than that derived from any of the designs previously discussed because of greater control over the complex meteorologic, hydrologic, and temporal factors. Although this design compares only a treated drainage with an untreated drainage, the results can be interpreted to indicate that the BMP's have improved water quality in the treated subbasins relative to the condition that would have existed without treatment. It should be noted that this design documents water quality improvements only in the treated subbasins; the accuracy of extrapolating results from the test basins to other portions of the project areas will remain untested. This experimental design may develop from a project area by chance, as BMP implementation progresses in subbasins with varying levels of success.

SUMMARY

For documenting water quality improvements resulting from BMP's within the shortest possible time period we believe the paired watershed design is clearly superior, because of its control of meteorologically-related variables. To document the magnitude of nonpoint sources prior to implementing BMP's, the above and below design

has advantages over the other designs. The before and after design is often the easiest design to follow, and can yield useful results provided that streamflows or some other surrogate measure of meteorologic variability is incorporated. Without correction for flow variability, it is unlikely that the before and after design can document BMP effects at the watershed level within any practical program time frame. It should be noted that for many of the experimental designs the time period required to observe BMP-related changes will depend upon how large a change is actually being made. For example, a 30 percent concentration reduction will take much longer to observe above the noise (variability) of the system than will a 90 percent reduction.

At least one of these experimental designs should be evident in any nonpoint source control project with water quality monitoring. The most appropriate monitoring strategies may include more than one of these experimental designs. The choice of the most appropriate design will depend upon the nature of the water resource impair-

ment, the water quality objectives of the project, the anticipated level and timing of land treatment, the topography of the project area, and the financial resources available for monitoring.

REFERENCES

- Baker, D.B. 1985. Regional water quality impacts of intensive row crop agriculture: A Lake Erie basin case study. *J. Soil Water Conserv.* 40(1): 125-32.
- Froehlich, H.A. 1976. Inorganic Pollution from Forests and Rangelands. *Water Resour. Res. Inst. Publ. No. Seminar-WR021-76.* Oregon State Univ., Corvallis.
- Johnson, C.W., et al. 1974. Sediment yield from Southwest Idaho rangeland watersheds. *Am. Soc. Civil Eng. No. 74-2505.*
- McCool, D.K., and R.I. Papendick. 1975. Variation of suspended sediment load in the Palouse Region of the Northwest. *Am. Soc. Civil Eng. Pap. No. 75-2510.*

MONITORING FOR WATER QUALITY OBJECTIVES IN RESPONSE TO NONPOINT SOURCE POLLUTION

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ABSTRACT

The broad application and continued use of herbicides and pesticides have resulted in a major diffuse source input of toxic material into aquatic ecosystems. The most common present water quality assessment practices cannot account for the entry of such compounds into the environment. Unlike point source input, where levels, quantity, and consistency of loadings are known, diffuse source input must be estimated using assessment procedures. Consequently, these assessment practices are not designed for the development or site-adaptation of water quality objectives. In Canada, water quality objectives are used for determining best land use practice and providing protection to the aquatic ecosystem. These demands on environmental assessments and the subsequent development of relevant water quality objectives can only be achieved by studies that provide insight into aquatic system behavior. The different environmental processes and fates that potentially regulate a compound's effect in the aquatic ecosystem emphasize the need for system behavior information. Examples from different systems illustrate the need for more comprehensive water quality assessment procedures to develop water quality objectives relevant to diffuse source inputs.

INTRODUCTION

Management of water quality within an aquatic ecosystem involves three facets: measurement, evaluation, and remedial action. Measurement pertains to the collection of physical, biological, and chemical water quality data. Evaluation necessitates a set of criteria with which the measured water quality can be realistically compared (in Canada these criteria take the form of water quality objectives—negotiated limits designed to protect and support designated water uses). These objectives provide the link between water quality information and the water uses to be protected and maintained within a given waterbody. Remedial actions, if required for use protection, are based on the measurement and evaluation information.

Approaches to the monitoring and assessment of water quality, as well as water quality management, will vary according to the relative significance of nonpoint and point source pollution. Specific water quality objectives that are used for evaluation do not vary similarly; however, their effectiveness depends upon the related data as well as the resultant management responses to them. Although water quality objectives have application for both nonpoint and point source pollution, the development of these objectives, their monitoring requirements, and the appropriate management strategies may differ significantly. The remainder of this discussion focuses on monitoring ap-

proaches, with regard to water quality objectives, and the assessment of nonpoint source pollution.

Point source inputs to an aquatic ecosystem are usually a consistent load of a given set of materials or chemicals. Data sets can be generated in specific areas of a river basin, and areas of noncompliance established with remedial actions confined to specific sources. Diffuse loadings from land use or atmospheric inputs tend to be more event oriented without a quantifiable area of effect in the aquatic environment. Basically, the complex nature of diffuse-source inputs results in the need for more comprehensive and extensive measurements and evaluation for development of a suitable management strategy.

Nonpoint source pollution in Canada most often results from agricultural practices, urban runoff, and atmospheric deposition. Aspects of these concerns are contained in three highly interrelated departmental priorities recently identified by Environment Canada: Toxic Chemicals, Long Range Transport of Airborne Pollutants, and Water Management (Environ. Can. 1983). To address these priorities, data must be assembled, an evaluatory mechanism initiated, and response programs implemented. Data collection necessitates an effective monitoring program; evaluation may correspond to the use of water quality objectives and the response usually consists of developing and implementing management options.

This discussion critically examines the measurements and evaluations required to develop water quality objectives specifically for nonpoint source inputs. Selected examples illustrate how such measurement might be used to determine the need and type of remedial action required to protect the aquatic environment.

MONITORING NONPOINT SOURCE POLLUTION

Water quality management requires a multiplicity of data to resolve the conflict of economic uses (industrial, agricultural) of water and the health of the aquatic environment (drinking water, fisheries, recreation). Historically, monitoring programs have been expected to yield information on many different aspects of water quality, and as a result data bases were established with many distinct and often incompatible rationales and designs. Generally, however, these measurement rationales and designs can be described in one of the following categories of environmental monitoring: (1) crisis response, (2) general monitoring, and (3) understanding aquatic processes.

Crisis Monitoring

Crisis monitoring, the oldest form of environmental data collection, includes observations such as the collapse of

certain fisheries in the Great Lakes, loss of potable water supply because of an epidemic such as the typhoid outbreaks that took place at the turn of the century, or the number of beach closures occurring over a certain period of time as happened on the Ottawa River. Although criteria information indicates a need for environmental management, it does not help make decisions to avoid such situations or identify solutions to ameliorate the problem; therefore, it is not relevant to this discussion.

General Monitoring

To determine the state of the aquatic resource requires a general monitoring program that will yield data describing the presence, level, and change over time of specific chemicals entering the aquatic system as a result of man's activities. Such programs include the collection of water samples at regular intervals and usually over the course of a number of years. Data generated from these collections are used to describe an average water quality condition of the sites. An example of such a network has been the general water quality monitoring carried out by the Water Quality Branch of Environment Canada, which is based on fixed sampling sites and monthly sampling frequencies (Whitow, 1985). Such a network emphasizes statistics to quantify the accuracy and precision of the baseline data generated (see Loftis et al. 1983; Sanders and Ward, 1978). When operated over a period of time, the program yields data suitable for long-term trend or intervention analyses.

Often, these data sets are also used to assess compliance with water quality objectives. Usually, these objectives are a simple concentration of a chemical in water, and the linking of the measurement and evaluation components of water quality management becomes little more than asking the perennial question, do ambient conditions comply with the objective? Because of our present reliance on fixed monitoring sites, considerable effort has been made to study the stochastic nature of general monitoring (Ward and Loftis, 1983) and determine the probability of exceeding a water quality objective or guideline at any particular point in time.

Compliance monitoring for water quality objectives in the Prairie Provinces is based on a two-level approach that provides a short- and long-term objective for each water quality variable of concern. The short-term objective is most commonly based on laboratory-derived criteria, whereas the long-term objective is developed from system variability (historic mean concentration ± 2 Standard Deviations) to account for seasonal variations. Considering the episodic nature of diffuse source loadings, the long-term objective is more relevant for water quality management concerned with diffuse source inputs. For example, the use of herbicides and pesticides in the Prairie Provinces follows crop cycles; application and land runoff provide event-oriented inputs to the aquatic ecosystem. General water quality monitoring in the area has demonstrated the presence and levels of pesticides throughout the area and indicated some presence of lindane and alpha-BHC in locations well beyond the areas of use (Gummer, 1978). Although such a data set indicates the need for water quality objectives, it does not provide the information to site-adapt the objectives with respect to potential effects within the system.

Process Assessments

Designing environmental monitoring or assessment to provide scientific advice for a specific issue requires a third type of assessment—monitoring to characterize the behavior of the system. Specific questions must be addressed. Is the correct substrate being sampled? Is the

hydrological regime of the system being taken into account? Are seasonal variations in concentrations and loadings being considered? These exemplify the need for a comprehensive multi-media approach to characterizing a system, if effective water quality objectives are to be developed and used to provide advice for sound water quality management. This requires a knowledge of the natural processes that regulate and often determine environmental quality within an aquatic system.

Environmental priorities such as acid rain or toxic substances make it critical to know both the environmental exposure and ecological effect of toxic chemicals. Exposure is a function of partitioning a chemical among the media under consideration (see Chapman et al. 1982); whereas, the effect is a function of the system's tolerance to the imposed stress. The need for process assessments was emphasized by Chapman et al. (1982). They concluded that a full understanding of the behavior of priority pollutants in the aquatic environment will require collecting considerably more information than chemical concentration in certain compartments.

By virtue of its diffuse nature, understanding of nonpoint source pollution relies more on monitoring and assessment than does point source pollution. Direct measurement of diffuse pollution sources is very difficult if not impossible; thus evaluation (using water quality objectives) depends upon a more careful monitoring of the system. General monitoring is often satisfactory for point source pollution because what and how much has been contributed to the system is known. However, without the benefit of accurate information on pollution inputs, more comprehensive monitoring is needed to evaluate nonpoint source pollution.

Process assessment requires measuring the system's variability and examining the physical-chemical and biological processes that determine environmental quality. Variability should consider statistical estimates of variance as well as include the comparison and analysis of the different sets of physical-chemical conditions. Understanding system behavior is an essential component of environmental management, and criteria, guidelines, or water quality objectives developed for good management practice must be adapted to system behavior. Process assessments provide the third step in developing relevant water quality objectives and implementing wise environmental management.

The value of process assessments is perhaps best described in the Great Lakes phosphorus management program. General monitoring provided estimates of total phosphorus loads within the lakes. From 1972 to the present, phosphorus loadings declined dramatically because of point source controls (1 mg/L), legislative controls (detergents), and nonpoint source controls (no till). To effectively manage phosphorus, and thus control the eutrophication of the Great Lakes, it was essential to determine what forms of phosphorus were most bioavailable and what sources should be emphasized for control programs.

Although it showed decreased loadings and concentration declines in Total Phosphorus, general monitoring could not provide the essential data to make such decisions. Process monitoring, such as bioassays of phosphorus availability and utilization, could distinguish the importance of the various sources. Consequently, appropriate decisions to target phosphorus loads for each of the lakes were made and agreement was reached on the most effective way to achieve the target levels.

During the 1960's, insecticides such as DDT and Dieldrin represented a major diffuse source input into Lake Michigan. Following the ban on the use and manufacture of these compounds in 1970, greater than 90 per-

cent declines of DDT levels were measured in bloater chubs between 1970 and 1980, and concentrations approached the Great Lakes Water Quality Agreement Objective of $1.0 \mu\text{g/L}$. Dieldrin, however, increased in bloater chubs over this time period, and concentrations continue to remain over the water quality objective of $0.3 \mu\text{g/L}$.

The different environmental behavior of these two compounds following regulatory action emphasizes the need for process information. When developing water quality objectives it is essential to know if a specific water quality objective is achievable and how long it might take to meet this objective. A lack of diffuse source input information makes it difficult to discern if further controls are required. What is the process that regulates levels of dieldrin in the environment, and why is it different from DDT? Process information is not yet available but is essential to answer such a question.

For the Great Lakes, water quality objectives supported by general monitoring have helped determine the need, type, and priority of remedial effort required. They provided an indication of the general health and response of the system. However, to maximize the effectiveness of water quality objectives, both in terms of their validity and especially their management potential, process information has been needed. Process assessments better resolve how to obtain the specific levels represented by the water quality objectives. They also evaluate the significance of nonpoint sources of pollution to encourage more efficient water quality management.

This point became apparent during the 1970's general monitoring programs in the Qu'Appelle River Basin of Saskatchewan which revealed that Province of Saskatchewan water quality objectives (which are not site-specific) for nutrients were routinely being exceeded. On the basis of this monitoring and evaluation, it was assumed that point source pollution was primarily responsible for this situation. Management adopted the position that controlling point source pollution would alleviate the problem. Tertiary waste treatment for the upstream cities of Regina and Moose Jaw was installed. Subsequent monitoring revealed little difference in nutrient values and it was not until detailed process assessments took place that a significant source of nutrients was determined to be of nonpoint origin. Present water quality objectives, which are not site-specific, have limited potential for water quality management because of the overall significance of nonpoint contribution of nutrients to the system. Therefore, process assessment in this case indicates that water quality objectives are probably not achievable through point source controls but require comprehensive nonpoint source mitigative measures.

CONCLUSIONS

Water quality monitoring for nonpoint source pollution must be taken into account for developing and maintain-

ing water quality objectives. Evaluating the significance of this pollution (through water quality objectives) and formulating management responses rely on more dynamic assessments than those provided by general monitoring. In some situations general monitoring that relies on describing average condition may be the most cost-effective solution to supporting (as opposed to developing) water quality objectives. However, nonpoint source pollution more often demands a detailed characterization of a system's water quality. A knowledge of the processes and interrelationships that regulate environmental quality within any aquatic system is usually required.

With Great Lakes eutrophication, the system forgave the immediate lack of process information. However, present day priority issues such as acid rain and the entry of toxic substances into the environment will not be as forgiving. In the interests of protecting and sustaining Canadian water resources, studies emphasizing the knowledge of the system must be carried out. These data are essential in developing relevant water quality objectives and designing networks to support them. Furthermore, as the use of water quality objectives grows, the information gained from such studies will provide some interpretable data, both in terms of information on the system and in assessing the health of these resources.

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REFERENCES

- Chapman, P.M., G.P. Romberg, and G.A. Vigers. 1982. Design of monitoring studies for priority pollutants. *J. Water Pollut. Control Fed.* 54(2): 292-7.
- Environment Canada. 1983. Annual Report 1982-83. Ottawa.
- Gummer, W.D. 1978. Pesticide monitoring in the prairies of Western Canada. *Water Qual. Inter. Rep. No. 4. Inland Waters Director., Environ. Can., Regina.*
- Loftis, J.C., R.C. Ward, and G.M. Smillie. 1983. Statistical models for water quality regulation. *J. Water Pollut. Control Fed.* 55(8): 1098-1104.
- Sanders, T.G., and R.C. Ward. 1978. Relating stream standards to regulatory water quality monitoring practices. *In Proc. Am. Water Resour. Assn. Symp. Establishment of Water Quality Monitoring Programs.* San Francisco.
- Ward, R.C., and J.C. Loftis. 1983. Incorporating the stochastic nature of water quality into management. *J. Water Pollut. Control Fed.* 55(4): 408-13.
- Whitlow, S.H. 1985. Water quality assessment in Canada. *Water Qual. Bull.* 10(2): 75-9.

USE OF BIOASSAYS TO DETERMINE POTENTIAL TOXICITY EFFECTS OF ENVIRONMENTAL POLLUTANTS

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ABSTRACT

Nonpoint source (NPS) runoff from mining, landfills, roads, croplands, grazing lands, and forests can contain chemicals harmful to aquatic organisms. Full scale biological surveys to determine their effects are difficult and costly. Bioassays of environmental samples integrate the effects of all toxicants contained in a sample. Biological organisms are being used more frequently to identify toxicant problems and to rank-order their severity. The Corvallis Environmental Research Laboratory (CERL) has developed a multi-media (aquatic/terrestrial) bioassessment protocol to assist in the identification of toxicity potentials associated with waste disposal. Similar techniques can be used to identify NPS pollutants. The bioassay response indicators are particularly useful in identification of field-site problems where complex mixtures of pollutants might be present. Use of the bioassessment protocol reduces the initial need for extensive chemical analyses, and produces data (toxicity LC_{50} information) in a form more readily understood by the public than bulk chemical concentrations. The CERL protocol has been used successfully to: (1) define and rank-order the effects of selected heavy metals, herbicides, and insecticides on microbes, earthworms, plant seeds, algae, daphnia, and fathead minnow larvae; (2) determine that rank-order of sensitivity differs with major toxicant groups; (3) detect the presence of bioactive organic and heavy metal mixtures in field site samples when concentrations of priority organic pollutants did not exceed EPA criteria levels; and (4) identify the basic chemical component of complex waste mixtures which produce environmental toxicological effects. These types of information should be useful in determining the potential effects of NPS pollutants and in designing measures for their control.

INTRODUCTION

Nonpoint sources (NPS) pollution problems are among the most pervasive, persistent, and diverse water quality problems facing the nation. This presents a definite problem to water quality decisionmakers who traditionally have addressed individual pollutants or site-specific sources of pollutants. The individual chemical-by-chemical approach requires a great deal of patience, time, money, and intellect to determine the pollutants adverse impact. Also, determining the substance producing the impact, the source of the substances, and the areal extent of the problem is difficult to address. Even extensive effort on a chemical-by-chemical basis does not assure an accurate ecotoxicological assessment, since one still has to relate environmental chemical measurements to biological/ecological impact. The approach most commonly employed is that of calculating potential toxicity based on chemical concentration of the 129 EPA consent decree chemicals (priority

pollutants) (Keith and Telliard, 1979) with extrapolation to water quality criteria. The approach has been useful in providing relative toxicity guidance, i.e., the relative toxicity of various chemicals under laboratory conditions. However, it has become increasingly apparent that this approach has severe limitations concerning realistic and accurate ecotoxicity estimates. Some of the problems associated with calculation of toxicity potentials based on priority pollutant chemical concentrations are that:

1. The data bases for most chemicals are not complete enough to permit the development of reliable criteria;

2. Most of the chemicals for which complete criteria exist are not necessarily those most commonly found in complex NPS or waste site discharges;

3. Application of criteria to field situations usually results in highly conservative and, therefore, overly restrictive estimates of toxicity or misinterpretation of toxicity cause and effect relationships;

4. Criteria for single chemicals were not intended to be assembled additively and there is little evidence to support that use; and

5. For contaminated soil and sediment there are no criteria on which to base decisions for judging if a site constitutes a problem.

Biological assessment of environmental toxicity alleviates most concerns associated with the above problems and provides a direct indication of potential toxicity (Roop and Hunsaker, 1985). An example was cited by Samoiloff et al. (1983) when they discovered that the most toxic sediment samples were those containing none of the EPA consent decree chemicals. Miller et al. (1985) have demonstrated similar results with the bioassessment of hazardous waste site samples using a multimedia bioassay procedure. Brown et al. (1984) demonstrated the inability of chemical analyses to provide a comprehensive evaluation of the toxicity potential of hazardous industrial wastes. They demonstrated further that a combined testing protocol using bioassays and organic chemical analysis was effective in identifying the toxicity potential of such wastes. A recommendation from Brown et al. (1984) was that a battery of bioassays be used to define the toxicity of wastes. The purpose of this paper is to demonstrate that such a bioassay test battery and analysis of results can be used to (1) identify and rank-order toxicity hazard potential of waste site samples; (2) help define and quantify areal extent of toxicity potentials; (3) help identify what chemical fractions of a complex waste contribute significantly to their overall toxicity; and (4) suggest that similar procedures might be used to assess the impacts of a broad spectrum of NPS pollutants. This paper is based, in part, on recently published and ongoing research conducted or sponsored by the Hazardous Materials Assessment Team at the EPA Corvallis Environmental Research Laboratory.

METHODS

Biological organisms respond to the adverse effects of a variety of specific pollutants (Fed. Water Pollut. Control Admin. 1968; U.S. Environ. Prot. Agency, 1976). However, there has been relatively little comparative toxicology done on environmental samples using a broad spectrum of organisms comprising both aquatic and terrestrial compartments of the ecosystem. For this purpose, we have adopted a multimedia bioassessment protocol described by Porcella (1983). The bioassays in the Porcella protocol include assessments of water and soil leachate toxicity on seed germination/root elongation (lettuce, *Lactuca sativa* L.), earthworms (*Eisenia foetida*), algae (*Selenastrum capricornutum*), daphnia (*Daphnia magna*), and fathead minnow larvae (*Pimephales promelas*). In addition, we have conducted Microtox (*Photobacterium phosphoreum*) tests (Beckman, 1982). Our approach has been to conduct comparative toxicological studies on pure chemicals and mixtures of chemicals in the laboratory to increase our confidence that biological responses to these substances are predictable and relatable to environmental samples (Miller et al. 1985). All toxicity responses are expressed as EC₅₀ or LC₅₀ concentrations for comparison.

We have focused on substances in chemical extraction groupings. Metals, base neutral organics, acid organics, and pesticides were extracted with water (4 ml water to 1 g soil). Bioassays were performed using these aqueous extracts. The predicted bioassay response, based on chemical concentration and criteria for certain chemicals, was then compared with bioassay responses on environmental samples dominated by the mixture of chemicals in question. This approach has permitted us to test the hypothesis that bioassay of environmental samples will produce EC₅₀ or LC₅₀ estimates significantly different from those predicted by calculation based on chemical concentrations with extrapolation to water quality criteria. Also, we have examined the relative toxicity potential of various metals, priority organics, and nonpriority organics in samples, from the Western Processing Superfund site at Kent, Washington. This was accomplished by incremental inactivation of metals with EDTA (at an EDTA to metals molar ratio of 4:1, based on Cu inactivation) and methylene chloride extraction of priority organic chemicals (Eichelberger et al. 1983) followed by algal assay examination. Chemical quality control was assured by surrogate spike recovery analysis coupled with daily calibration of the GC/MS system.

Extent of chemical contamination was determined using a modified phytotoxicity test described by Thomas and Cline (1985). Lettuce seeds were used to test the toxicity potential of soils collected along four 90 m long parallel transects that were 15 m apart. Soils from 0-15 cm depth and 15-30 cm depth were used since they encompassed the root zone in the area. The site was located downwind, along a suspected concentration gradient perpendicular to an open ditch known to have transported liquid organic wastes associated with the manufacture of herbicides, insecticides, and neurotoxin gases at Rocky Mountain Arsenal, Colorado. Thomas et al. (1984) have described the statistical sampling design in greater detail. Phytotoxicity data from the site were analyzed using kriging. Kriging is a statistical technique developed in the mining industry (Clark, 1982). Only a limited number of samples are required to successfully define a contaminated area using kriging. The technique employs a weighted moving average that calculates point estimates or block averages over a specified grid. Output of the kriging analysis for this study is a contour map displaying areal variation in phytotoxicity.

FINDINGS AND DISCUSSION

Comparative Toxicology

Miller et al. (1985) conducted comparative toxicological studies on several known single and complex organic and metal contaminants in the laboratory using the Porcella (1983) bioassessment protocol plus the Microtox Test (Beckman, 1982). They concluded that:

1. The protocol test organisms responded differentially to various pollutants and their EC₅₀ or LC₅₀ results generally conformed to the range of values reported in the literature for individual chemicals and metals;
2. Test organism rank order of sensitivity differed with major toxicant groups, suggesting that certain bioassays are better suited than others to assess given chemical groups;
3. Algae (*Selenastrum capricornutum*) was the most uniformly sensitive test organism across a broad spectrum of pollutant groupings; and
4. Differences in sensitivity levels of the test organisms, relative to the toxicant assayed, can be used to identify those biotic components most susceptible to the presence of toxicants and to draw an educated conclusion as to the contaminant type producing the toxic effect.

Based on the conclusions drawn from bioassay responses to pure chemical substances in the laboratory and the assumption that bioassays integrate the toxicity effects of all sample components regardless of their composition, Miller et al. (1985) bioassayed soil and soil elutriate samples from seven diverse hazardous waste sites (Table 1). The samples were dominated by heavy metals, solvents, phthalates, phenols, pesticides, and herbicides.

Relative, integrated biotic toxicity of the sites and their rank ordering could be determined by calculating the arithmetic average toxicity across the different tests in Table 1. If one was concerned primarily with potential aquatic impacts, the algae, Daphnia, and Microtox tests probably would be the most applicable indicators. The sensitivity of algae appears to be much greater than the other bioassays for most of the samples.

Algae responded adversely to all but one of the samples. In that case, no aquatic test responded adversely. Toxicity rank ordering, such as that shown in Table 1, would be helpful in: (1) determining potential environmental impacts; (2) directing further chemical analyses within sites; and (3) ranking cleanup across or within various sites. Bioassay data might be used to monitor toxicity changes in samples before and after waste cleanup or the adoption of various NPS management alternatives, thus helping to determine the degree of treatment success.

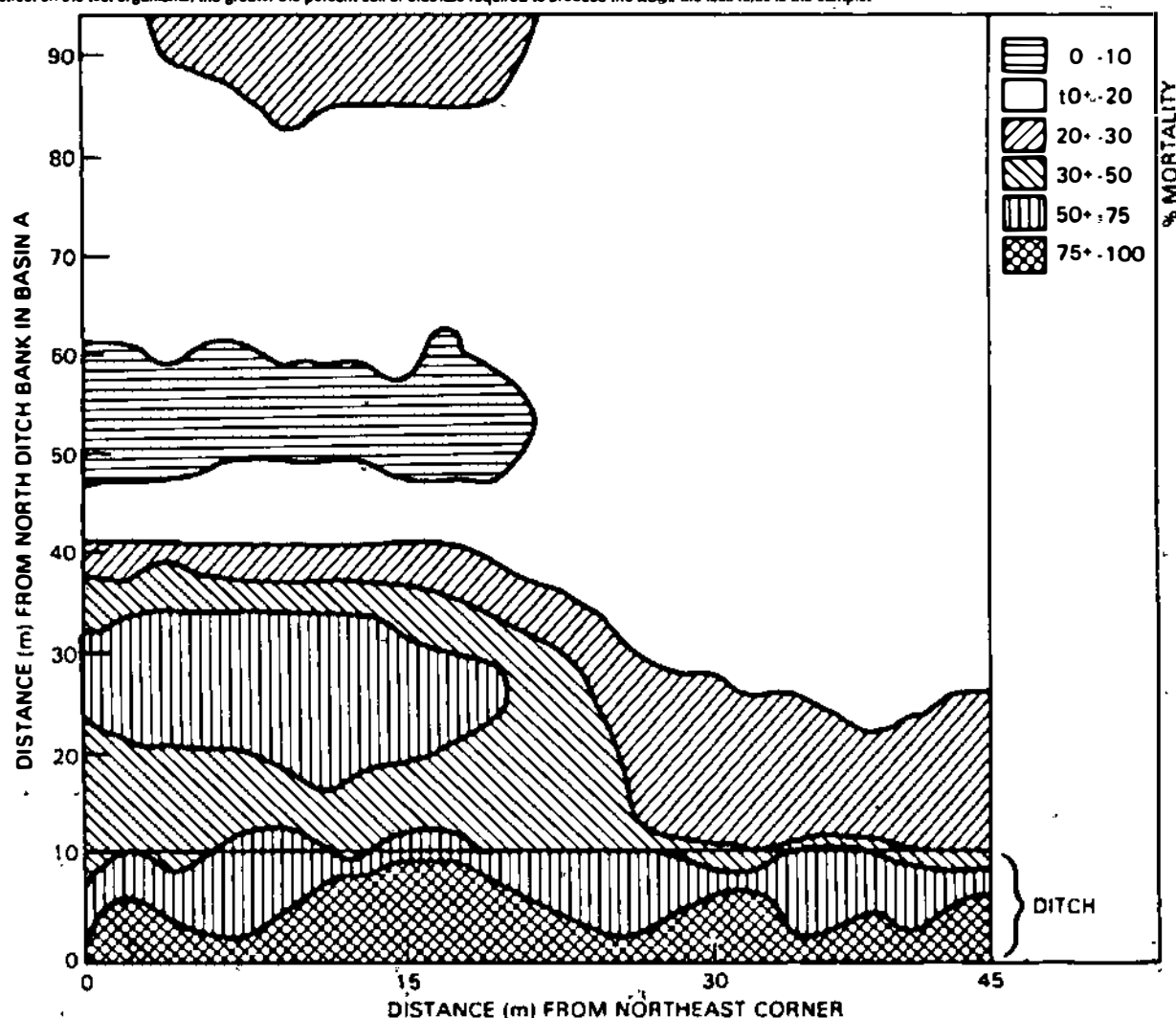
Kriging of Bioassay Data

Another means of assimilating bioassay data into a format useful for problem solving and remedial design relative to chemical hazard assessment is that of kriging. Phytoassay responses for soil samples from Rocky Mountain Arsenal were subjected to kriging as described under methods. Kriging the 0-15 cm phytotoxicity data, with the resultant toxicity potential contours is shown in Figure 1.

Thomas et al. (1984) compared kriged phytotoxicity bioassay estimates (Figure 1) with sample site-specific plant mortality data (Figure 2). This type of graphic interpolation could be very useful in making waste site cleanup decisions or in designing NPS watershed or ecoregional contaminant source controls. For example, if it was determined that the 30 percent mortality contour should be used as the criterion for remedial action for the conditions shown in Figure 2, the area below the 30 percent solid contour line would be targeted for action.

Table 1.—EC₅₀ response for percent in soil (earthworm) or soil elutriate with associated complex chemical contaminants from selected hazardous waste sites.

Waste Site	Major Chemical Group	Bioassay Response (Percent of soil or elutriate required to produce an EC ₅₀)					Arithmetic Average
		Algae	Daphnia	Microtox	RE ¹	Earth-Worm	
Holder Chemical West Virginia	Pesticides, herbicides	2.1	3.6	18.0	3.6	70.0	19.5
Western Processing Kent, WA #17	Heavy metals, phenols, solvents, pesticides	0.2	5.6	2.2	37.0	55.0	20.0
Big John Houldt West Virginia	PAH ² , unknown organics	5.4	87.0	28.0	NE ⁴	<10.0	46.1
Hollywood Memphis, TN	Pesticides	24.0	22.0	>90.0	NE	>25.0	52.2
Sharon Steel Fremont, NY	Heavy metals, tar, PAH	0.6	30.0	99.0	NE	>75.0	40.4
Sapp Battery Cottondale, FL	Heavy metals	41.0	70.0	NE	NE	NE	22.2
Thiokol Chester, WV	Diphenylamine	NE	NE	NE	NE	35.0	87.0

¹Root elongation test.²Earthworm 14 day soil contact test (LC₅₀).³PAH = polynuclear aromatic hydrocarbons.⁴NE = No effect observed at 100% of the soil or soil elutriate. Therefore, NE is factored into the arithmetic average as 100%, i.e., 100% of the soil or elutriate produced no effect on the test organisms; the greater the percent soil or elutriate required to produce the EC₅₀, the less toxic is the sample.**Figure 1.—Estimated lattice seed mortality (based on kriging) for the 0-15 cm soil fraction from the Rocky Mountain Arsenal (from Thomas et al. 1984).**

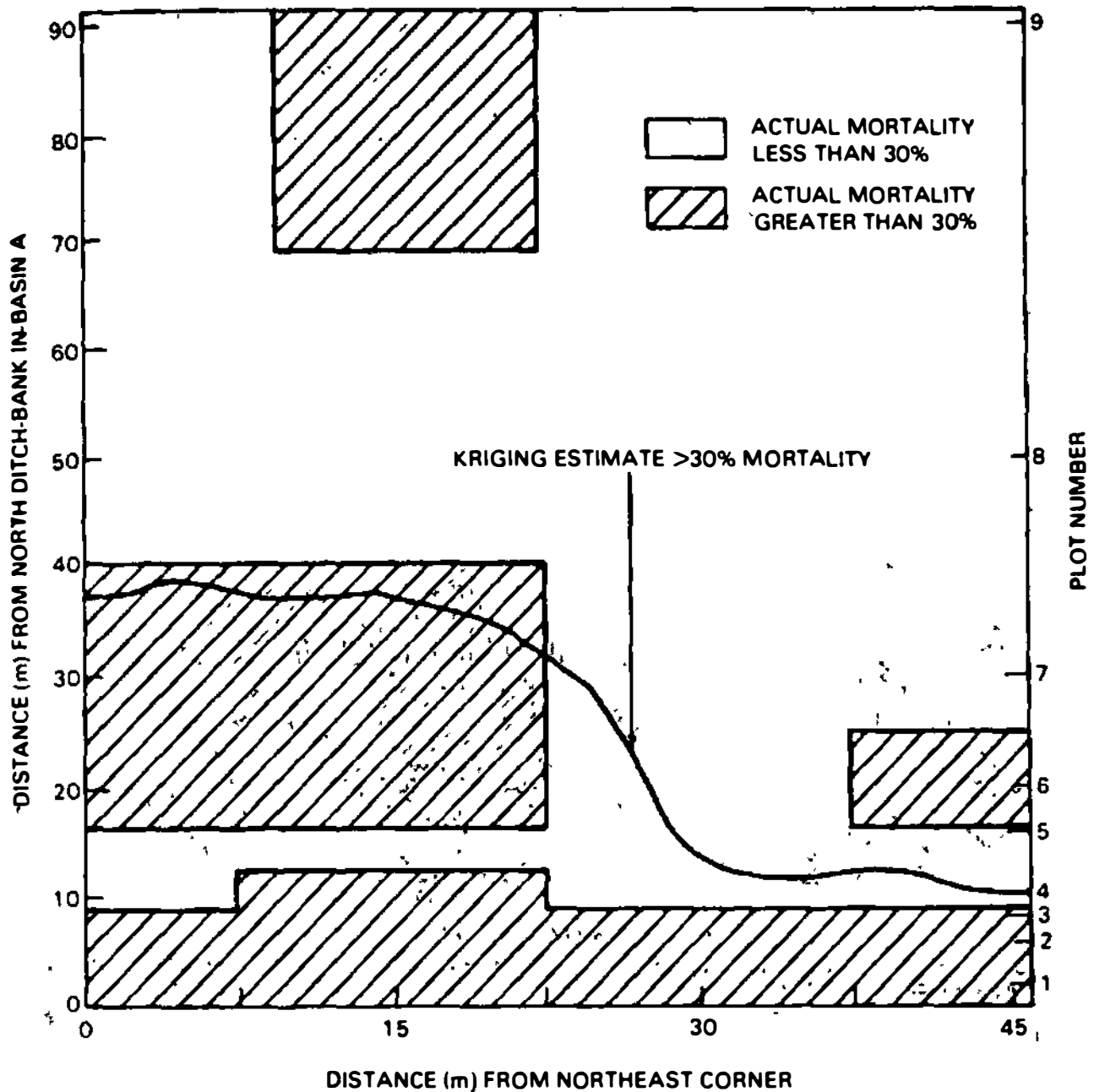


Figure 2.—A comparison of greater than 30 percent lettuce seed mortality (estimated from kriging) to observed lettuce seed mortality for the 0-15 cm soil fraction from the Rocky Mountain Arsenal (from Thomas et al. 1984).

Unfortunately, the hazardous waste site situation is more complex than the kriged phytotoxicity surface data (0-15 cm deep) would indicate. Samples from the 15-30 cm depth at the same site produced the results shown in Figure 3. Comparison of kriging estimates with plant mortality data at this depth is shown in Figure 4. It is evident that site cleanup based on the surface sample greater than 30-percent mortality results would omit significant areas of contamination. This information makes the remedial action plan more complicated, but it adds significant realism to the site assessment. A final remedial action decision that includes consideration of chemical bioavailability as determined by integrative bioassay endpoints should greatly enhance the probability of contaminant cleanup success. Chemical information alone cannot assure an accurate assessment of toxicity potentials and in some instances might lead to misinterpretation of toxicological cause and effect relationships.

Chemical Analysis and Bioassay

Hazardous waste assessment and NPS pollution problems are similar in that each has traditionally been assessed from a chemical perspective. Severity of the problem has been assessed relative to the concentration of a given chemical producing a given type and degree of response under laboratory conditions. Controlled condition laboratory response tests have been used extensively to develop water quality criteria for various chemicals. Problems associated with the extrapolation of these criteria to assess field conditions were mentioned in the introduction. In addition, combinations of pollutants and different attenuating characteristics of a site are difficult to assess when calculating toxicity estimates.

Direct bioassay of samples tends to minimize many of these problems. Bioassays integrate the toxicological effects of all sample components regardless of their type

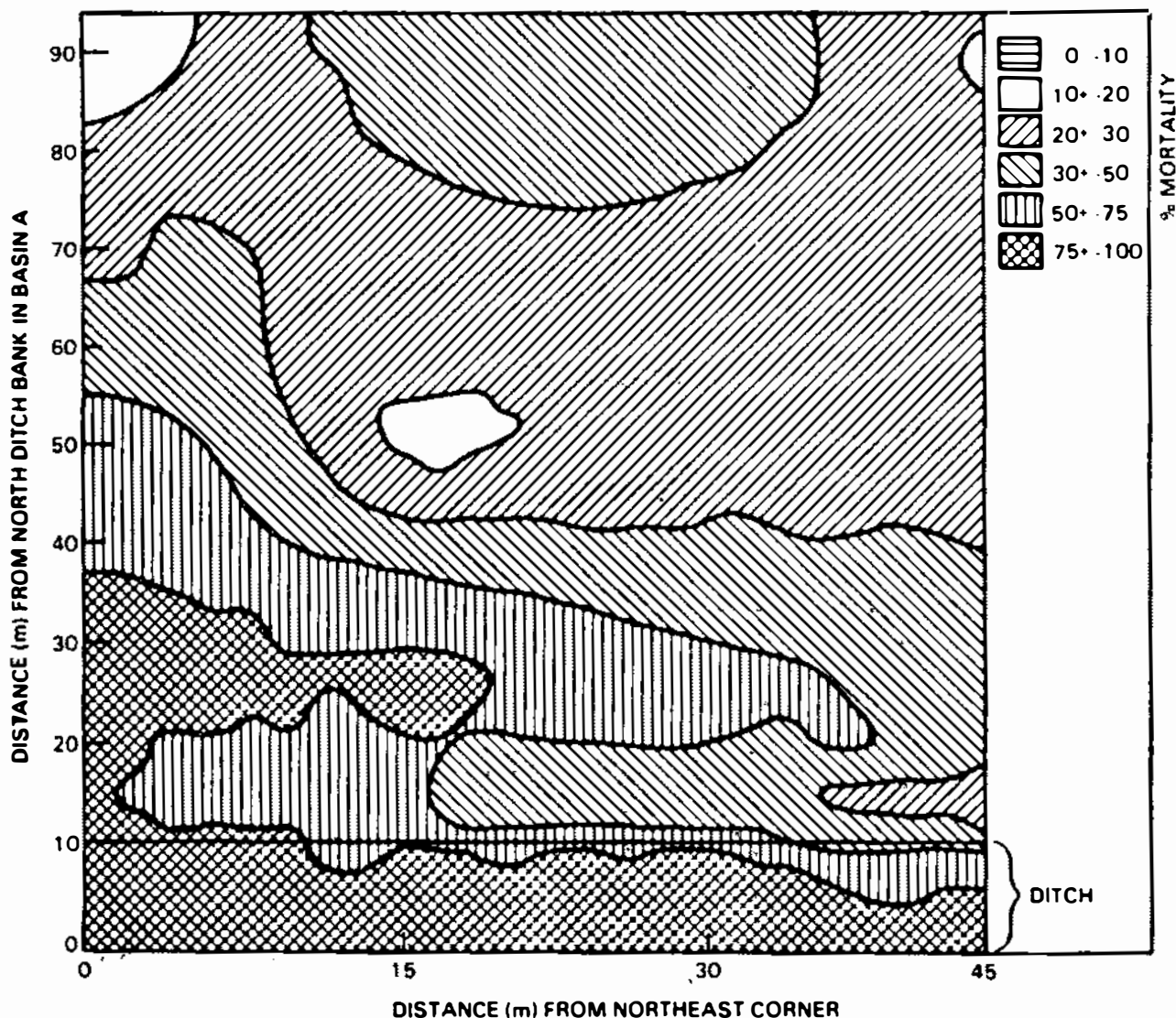


Figure 3.—Estimated lettuce seed mortality (based on kriging) for the 15-30 cm soil fraction from the Rocky Mountain Arsenal (from Thomas et al. 1984).

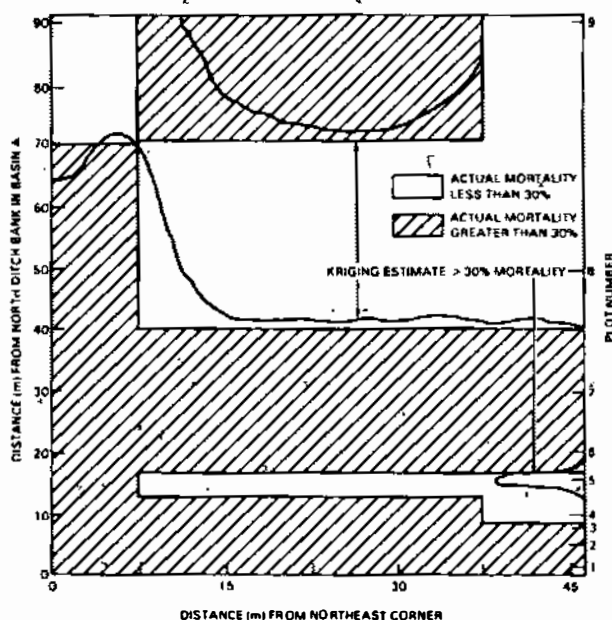


Figure 4.—A comparison of greater than 30 percent lettuce seed mortality (estimated from kriging) to observed lettuce seed mortality for the 15-30 cm soil fraction from the Rocky Mountain Arsenal (from Thomas et al. 1984).

and amount. Chemical presence is of limited concern, but bioavailability of the chemical and its effect on the test organism are of great concern. Bioassay of the waste samples provides a direct estimate of the chemical's toxicity potential.

Hazardous waste cleanup decisions have relied heavily on analysis of the EPA 1965 consent decree chemicals (the 129 so-called priority pollutants). Concentrations of these pollutants in excess of water quality criteria values have been used to justify various cleanups, but in many instances environmental criteria do not exist. In these cases the chemical information may be more misleading than it is helpful since one suspects there may be some hazard, but there is little information for determining the degree of hazard based on the chemical analysis.

Herein lies the benefit of the bioassay procedure. Soil and water bioassays in the Porcella (1983) bioassessment protocol will provide an indication of toxicity to various compartments of the system. Also, it will provide a quantitative (EC_{50} or LC_{50}) ranking of the toxicological impact potentials among those compartments.

We believe that reliance on chemical criteria alone, and particularly those for priority pollutants, could lead to erroneous decisions concerning remedial actions. The general chemical analytical protocol for hazardous waste site samples calls for priority metal and organic identification

Table 2.—Concentration (mg/L) of priority and nonpriority organics contained in sediment leachates from Western Processing (modified from Miller et al. 1985).

Constituent	Control	East Ditch Water	Pond 005	017	020
Phthalates	0.002	0.010	0.001	0.011	0.002
Ethylbenzene	0.002	<0.001	0.001	0.001	0.001
Nitrosamines	<0.001	<0.001	<0.001	0.010	<0.001
Phenol	<0.001	<0.001	<0.001	18.370	<0.001
Nonpriority Organics	0.3	15.9	0.8	11.9	0.2
Total Organics	0.3	15.9	0.8	29.9	0.2

Table 3.—EC₅₀ or LC₅₀ response in % sediment¹ (earthworm), sediment elutriate, and surface water to chemical contaminants in Western Processing Samples (modified from Miller et al. 1985).

Test Organism	East Ditch Control	Pond Water	005	017	020
Algae	45	0.8	0.4	24.9	NE ²
Daphnia	90	18.5	3.3	NE	NE
Microtox 5 min	NE	82.7	41.2	55.4	NE
15 min	NE	21.3	<5.6	50.1	NE
30 min	NE	10.2	<5.6	43.4	NE
Lettuce RE	NE	NE	61.4	49/100 ³	NE
Earthworm ⁴	NE	—	>50<100	>100	—
Mean EC ₅₀ ⁵	87.0	32.4	34.1	73.7	>100

¹ Sediment/soil samples.² NE = no significant toxicity was observed.³ 49/100 = 49% inhibition in 100% sediment elutriate.⁴ LC₅₀ values = concentration at which 50% mortality occurs.⁵ Mean of algae, daphnia, microtox 30 min, lettuce RE, and earthworm tests.

ganic extracted (methylene chloride) sample. Results show that chelation of soluble metals with EDTA decreased toxicity 90-fold, but that the chelated elutriate remained highly toxic (Figure 7). Significant additional toxicity reduction was not realized when the sample was subjected to combined chelation and priority organic extraction. It appears from this analysis that metal toxicity dominated the Western Processing samples, but that nonpriority organic chemicals alone were sufficient to classify the soil leachate as highly toxic. The toxicological influence of priority organics in these samples appears to have been minimal. Therefore, predicted toxicity of these samples based on the concentration of priority pollutants would have severely underestimated sample toxicity.

SUMMARY

We have attempted to develop a biological toxicity screening protocol that has broad-based application potential. Based on results to date we believe that

1. A modified Porcella bioassessment protocol can be used to define and rank order the effects of selected heavy metals, herbicides, and insecticides.

2. Selected segments of the protocol can be used to assess the influence of complex wastes under field conditions, i.e., there is a relationship between laboratory bioassay responses to environmental samples and actual field conditions.

3. The protocol can be used to assess environmental toxicity potentials in situations where water quality criteria are lacking or nonexistent.

4. Direct bioassay of environmental samples produces toxicity results significantly different from those predic-

tions based on measured chemical concentrations with extrapolation to water quality criteria.

5. Experience gained from the bioassay of hazardous waste site samples should have application to many aspects of the NPS pollution problem.

6. Algal assay appears to have great universal toxicant/stimulant assessment potential based on sensitivity to various toxicants.

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REFERENCES

- Beckman, Inc. 1982. Microtox System Operating Manual. Beckman Instruments, Inc. Microbics Operations. Carlsbad, California.
- Brown, K. W., K. C. Donnelly, and J.-C. Thomas. 1984. Use of short-term bioassays to evaluate environmental impact of land treatment of hazardous industrial waste. EPA-600/S2-84-135.
- Clark, I. 1982 Practical Geostatistics. Appl. Sci. Publ., London.
- Robert S. Kerr, Environmental Research Laboratory, Ada, Oklahoma 74820. Clark, I. 1982. Practical Geostatistics. Applied Sci. Publishers, London.
- Eichelberger, J. W., E. H. Kerns, P. Olynyk, and W. L. Budde. 1983. Precision and accuracy in the determination of organics in water by fused silica capillary column gas chromatography/mass spectrometry and packed column gas chromatography/mass spectrometry. Anal. Chem. 55:1471-79.

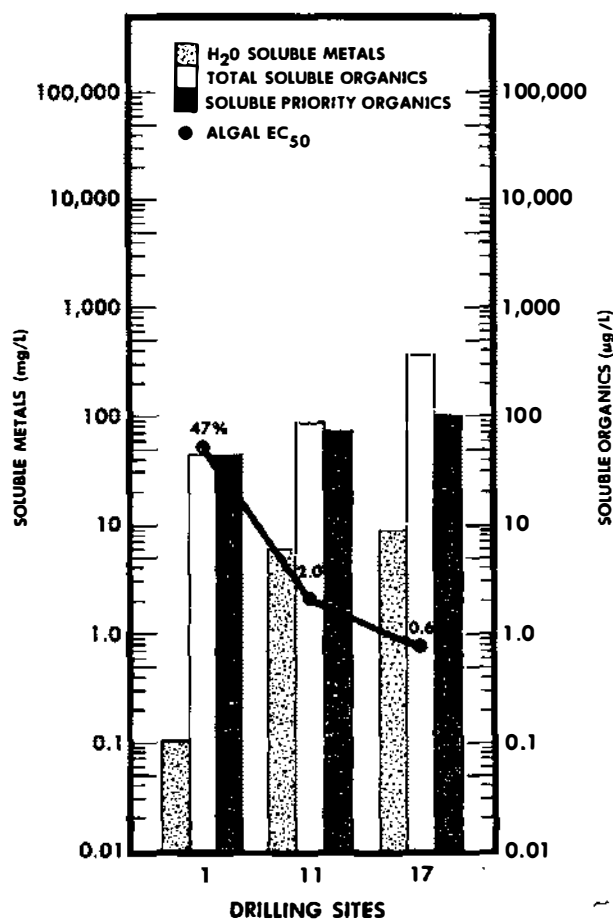


Figure 6.—Response of algal assay, EC₅₀ (*Selenastrum capricornutum*) to chemical contaminants in 0-1 m depth soil elutriates from well drilling site numbers 1, 11, and 17 at the Western Processing waste site.

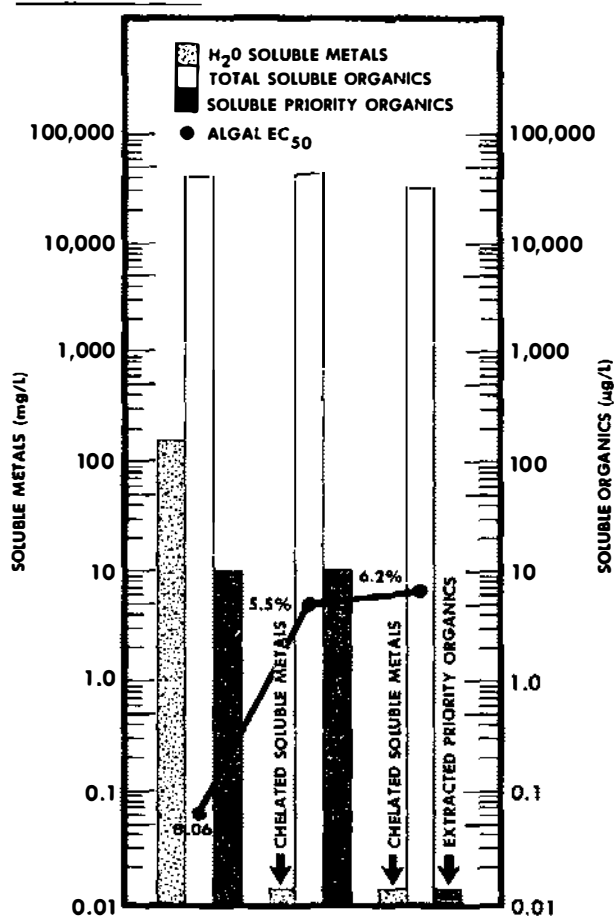


Figure 7.—Algal assay EC₅₀ response to Western Processing soil elutriate from well drilling site 17 at the 3-4 m depth prior to and after chelation of heavy metals and extraction of priority organic pollutants.

Federal Water Pollution Control Administration. 1968. Water Quality Criteria. Report of the National Technical Advisory Committee to the Secretary of the Interior. U.S. FWPCA, Washington DC.

Kéith, L. H., and W. A. Telliard. 1979. Priority pollutants: a perspective view. Environ. Sci. Technol. 13:416-23.

Miller, W. E., S. A. Peterson, J. C. Greene, and C. A. Callahan. 1985. Comparative toxicology of hazardous waste site bioassessment test organisms. Environ. Qual. (accepted for publication, May 1985).

Porcella, D. B. 1983. Protocol for Bioassessment of Hazardous Waste Sites. EPA-600/2-83-054 U.S. Environ. Prot. Agency, Corvallis, OR.

Roop, R. D., and C. T. Hunsaker. 1985. Biomonitoring for toxics

control in NPDES permitting. J. Water Pollut. Control Fed. 57:271-277.

Samoloff, M. R., et al. 1983. Combined bioassay-chemical fractionation scheme for the determination and ranking of toxic chemicals in sediments. Environ. Sci. Technol. 17:329-333.

Thomas, J. M., et al. 1984. Characterization of chemical waste site contamination and its extent using bioassays. Battelle PNL-5302. PNL, Richland, WA.

Thomas, J. A., and J. F. Cline. 1985. Modification of the Neubauer technique to assess toxicity of hazardous chemicals in soil. Environ. Tox. Chem. 4:201-207.

U.S. Environmental Protection Agency, 1976. Quality Criteria for Water. EPA-440/9-76-023. Washington, DC.